Seasonal prescribed burning impacts to northern Minnesota lowland brush ecosystem plant communities

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Abstract

Minnesota lowland brush ecosystems provide critical habitat for wide variety of wildlife including over 80 Species of Greatest Conservation Need. These ecosystems depend on fire disturbance to inhibit woody plant encroachment and maintain the herbaceous plant community. Without fire, woody plants become dominant in the overstory, reduce herbaceous cover, and reduce the quality of habitat for wildlife that rely on spatial and structural diversity. Natural resource managers use prescribed burning to decrease woody plant density and enhance the herbaceous plant community in these ecosystems. Currently, the prescribed fire regime in lowland brush ecosystems is largely limited to the spring season, and research in other ecosystems has found that burn season can result in a wide range of impacts to burn severity and both woody and herbaceous plant communities. Understanding the impacts of burn season is critical for natural resource managers to effectively conduct prescribed burns to meet their management objectives. However, little research currently exists on the impacts of fire in lowland brush ecosystems, let alone burn season.

The objective of our study was to determine whether season of prescribed burning led to significant differences in burn severity, changes in total woody plant density, changes in density of common woody species, and changes in herbaceous cover. In 2016, we established permanent sample plots at 4 study sites throughout northern Minnesota in order to monitor burn severity and response of vegetation. Each site was broken into 4 burn units including a spring, summer, fall, and a control where no burn was conducted. The results of this study focused on the impacts of burn season on burn severity and the

plant community in the first growing season after the burns were conducted, and includes the results of four spring burns, two fall burns, and two summer burns.

While we found that burn severity and plant community response differed among burn season treatments, our findings varied by spatial scale. Burns conducted in the spring burned more area those in fall or summer. Given that more area burned in spring compared to fall or summer, we examined the impacts of burn season at different scales to consider broad landscape scale impacts (burn unit scale) and direct fire impacts (plot scale). Burn unit scale included analysis of all permanent sample plots within each burn unit regardless of whether there was evidence of fire at the plot, and at the plot scale we included only on sample plots where evidence of fire was present. Additionally, we broke down the direct fire impacts at the plot scale to look at impacts on common woody plant species and species groups. At the burn unit scale, spring burns were the most severe, resulted in the highest amounts of topkilled woody stems, and the overall greatest reduction in woody plant density even though vigorous resprouting was likely occurring. At the plot scale, burn severity did not differ among burn seasons, but spring burns still resulted in an overall reduction in woody stems while fall and summer burns did not. Furthermore, woody species varied in their response to burn season with some species appearing to resprout prolifically and others not as much, while herbaceous cover did not change as a result of fire compared to the control units.

Our results indicate that spring burns were the most successful at reducing woody stem density one year after burn. However, reduction in woody stem density may not be the only management objective. Our results also suggest that spring burns create a uniform understory of shrub regeneration, which may reduce heterogeneity on the

landscape. Recent research suggests that high severity burns, which create a single cohort of regenerating woody shrubs, reduce habitat quality for the bird community (Zlonis et al., 2019). Thus, natural resource managers should view fire season as a tool for supporting a variety of outcomes in lowland brush ecosystems.

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Introduction

Fire disturbance plays a pivotal role in the conservation of wetland and grassland ecosystems worldwide (DeBano et al., 1998; Nowacki & Abrams, 2008; Whelan, 1995). Fire increases habitat heterogeneity supporting higher abundances of plant and wildlife species (Franklin et al., 2007; Lyon et al., 2000), promotes herbaceous plant communities, and inhibits shrub encroachment (Bart et al., 2016; Curtis, 1959; Vogl, 1969). In the absence of fire, woody plant abundance increases and can lead to reduced habitat quality for wildlife associated with open wetland habitat such as sharp-tailed grouse (Hanowski et al., 1999, 2000), reduced diversity within the plant community (Ratajczak et al., 2012), and conditions unsuitable for carrying future fires (Scheffer et al., 2001). Historically, fire disturbance was common and maintained wetland and grassland ecosystems in early-successional stages (Curtis, 1959; Vogl, 1969). However, following European settlement and agricultural expansion, fire was largely suppressed (DeBano et al., 1998; Vogl, 1974).

Today, the benefits of fire are widely recognized and prescribed burning is frequently used to manage fire dependent ecosystems (Nuckols et al., 2011; Rooney & Leach, 2010; Tix et al., 2003). In Minnesota, the Department of Natural Resources (MN DNR) uses prescribed burning to maintain early successional habitat for wildlife species strongly associated with open vegetation types such as lowland brush ecosystems (LBEs). Prescribed burns in these ecosystems are most frequently conducted in the spring (April-May) prior to the start of the growing season. This spring period provides ideal

conditions for burning for a number of reasons including: the presence of fully cured herbaceous vegetation, low fuel moisture, easily managed smoke conditions, and safety for burn crews (Knapp et al., 2009). While conducting prescribed burns in other seasons is less common, the historical occurrence of fire in these ecosystems may have been throughout the spring, summer, and fall due to either intentional fire setting by American Indians (Higgins, 1986), or a result of lightning strikes (Pyne, 1982; Zajac & Rutledge, 2001).

Research has found that conducting prescribed burns in different seasons leads to significant differences in species abundance and richness across different plant communities (Biondini et al., 1989; Copeland et al., 2002; De Groot & Wein, 2004; Knapp et al., 2009; Lovell et al., 1982; Steuter, 1987; Towne & Owensby, 1984; Viereck & Schandelmeier, 1980). A growing body of evidence also suggests that conducting prescribed burns or other brushland management techniques in the growing season, instead of the dormant season, may result in a reduction of vigor in resprouting woody plants after burn (Buckman, 1964; De Groot & Wein, 2004; Pelc et al., 2011; Ruthven et al., 2003). Given that MN DNR land managers use prescribed burning in LBEs to maintain early successional habitat, inhibiting shrub encroachment through growing season burns and reducing vigor of woody resprouts is an attractive option. Studies are now underway to determine the impacts of seasonal prescribed burning to LBE plant communities. However, another primary objective for MN DNR land managers is conducting high severity fires that result in a high proportion of aboveground woody stem mortality, hereafter called topkill. Aboveground stem mortality is frequently referred to as ‘topkill’ when the root systems of woody plants remain alive belowground

and only the aboveground stem is killed as a result of the fire. There are many factors that can influence the level of severity of a prescribed burn and these factors can all vary among seasons.

The severity of fire is determined by the impact a fire has on the surrounding vegetation (Keeley, 2009), and is a function of a fire's intensity, or direct energy exhibited by the flames during a fire, and how fast a fire moves through the landscape also called its rate of spread (Whelan, 1995). Fire intensity and rate of spread are both influenced by a variety of factors including the fuel types and sizes available, heterogeneity of fuels across the landscape, species flammability, fuel moisture, wind speed and relative humidity (Whelan, 1995). Fuel related factors, such as fuel moisture and fuel bed heterogeneity, fluctuate across seasons and are considered to have the greatest impact on fire intensity and rate of spread (DeBano et al., 1998; Scott & Burgan, 2005; Whelan, 1995). This may be particularly important in herbaceous layer fuel types where surface fires drive the impacts to the surrounding vegetation and level of fire severity (DeBano et al., 1998). Therefore, seasonal variation in the herbaceous fuel layers found within LBEs may result in differences in severity and the resulting topkill.

While research on prescribed burning in LBEs is limited, we can look at research in other surface fire systems to get a sense for the potential impacts of burn season on fire severity. Research from western ponderosa pine and northern mixed prairie ecosystems are two examples of how the season in which a fire occurs can impact the severity of the fire, but this research also suggests that results may vary by ecosystem because of seasonal variation in fuel loads (Kerns et al., 2006; Steuter, 1986). In western ponderosa pine forests, while the historical fire season was likely spring and fall when fuel loads

were at their driest, spring prescribed burning is commonly practiced because of more easily manageable fuel conditions (Agee, 1993). In order to understand the effects of burn season in this system, Kerns, Thies, & Niwa (2006), conducted prescribed burns in the spring and fall and found that prescribed burns conducted in the fall resulted in higher severity fires compared to the spring. These high-severity fall fires resulted in a greater percentage of trees with scorch around the bole and higher scorch, and ultimately higher tree mortality and a reduction in basal area (Kerns et al., 2006). While fall fires resulted in higher severity fires in western ponderosa pine forests, research in northern mixed prairies has found no difference in severity among fall and spring fires, but a significant difference in summer (Steuter, 1986). Steuter (1986) conducted a study on prescribed fire season and its impacts on fire intensity and rate of spread, key factors determining severity. The study found that these severity factors were greatly reduced during summer burns, but that there was no difference in these factors among spring and fall burns (Steuter, 1986). Ultimately, the reduction in fire intensity and rate of spread limited the spatial impact of burns during the summer season as fire was at times unable to carry through research plots (Biondini et al., 1989; Steuter, 1986). While both of these studies occurred in surface fire systems, where fire is carried through surface herbaceous fuels similar to LBEs, the burn season impacts to fire severity varied between the ecosystems.

Studies in other ecosystems have found that fire severity varies by season of prescribed burning, and fuel related factors fluctuate across season and in turn influence fire intensity and rate of spread. Therefore, our objective was to study how seasonal prescribed burning impacts fire severity within LBEs through analysis of topkill and occurrence of fire at sample plots. Given that our prescribed burns in the summer would

take place during peak growing season, we expected fuel conditions would limit fire intensity and rate of spread during summer burns. Therefore, we hypothesized burn season would have a significant association with the occurrence of fire at sample plots, and that plots in the spring and fall would be more likely to burn than plots in the summer. Additionally, given our expectation of spatial limitations in the occurrence of fire during the summer burns, we also hypothesized that burn season would have a significant effect on the amount of topkill, and that prescribed burns conducted in the spring and fall would exhibit higher topkill than those conducted in the summer.

Methods

Study system

The Minnesota lowland brush ecosystem is a diverse mix of plant communities ranging from alder swamp, shrub carr, ericaceous shrub fen, to wet meadow. Common wetland woody species of these plant communities include *Salix petiolaris* Sm., *Salix discolor* Muhl., *Salix pyrifolia* Andersson, *Cornus sericea* L., *Alnus incana* (L.) Moench *ssp. rugosa* (Du Roi) R.T. Clausen, *Rhododendron groenlandicum* Oeder, *Chamaedaphne calyculata* (L.) Moench, and *Betula pumila* L.. There is also a broad diversity of herbaceous wetland forbs, grasses and sedges. Trees are relatively a minor component of these plant communities, and if present, rarely develop into a mature overstory canopy. While these ecosystems include a wide diversity of plant species, they all share the common characteristic of a water table close to or above the soil surface for most of the year (MN DNR, 2003).

While Minnesota lowland brush ecosystems occur throughout the state, our study sites were located in northeast Minnesota. The climate of this region ranges from an average July temperature of 19.1°C to an average January temperature of -12.4°C, with a growing season that occurs from late-May to mid-September (NOAA, 2019). Precipitation of the northeastern Minnesota region is 67.4 cm annually (NOAA, 2019).

Study sites

Our study was conducted at four study sites located in northeastern Minnesota within Aitkin, Carlton, and St. Louis counties (Fig. 1-1). Sites included a mix of private and public lands designated as Priority Open Landscapes by the MN DNR. Each study site included 4 treatment units (control, spring, summer, fall), and each treatment unit was roughly 40 hectares. Between fall 2016 and spring 2018, a total of eight prescribed burns were completed at the four study sites and included four spring burns, two fall burns, and two summer burns (Table 1-1). All prescribed burns were completed by MN DNR prescribed burn crews. Spring burns occurred in mid-late May and represent the traditional dormant season prescribed burns that occur in the region for LBEs. Summer burns were carried out in the growing season during July – September, and fall burns in October and November (after the first-frost).

Data collection

To monitor the severity of prescribed burns in LBEs, permanent vegetation sample plots were established prior to burn treatments and vegetation surveys were completed both before and after burns. Vegetation sample plots were established for both

vegetation surveys and to support a collaborating study on the impact of burn season on the breeding bird community (Hawkinson, 2019). Methods for the breeding bird study required the establishment of eight point count locations to monitor bird activity. Point count locations were evenly distributed across each treatment unit, and at each point count location we established two vegetation sample plots (Fig. 1-2). Plot centers were marked with rebar posts that could withstand fire and ensure plots could be relocated in subsequent years. In sum, each treatment unit started with 16 vegetation sample plots, and vegetation surveys were conducted from July-August 2016, 2017, and 2018.

During vegetation surveys, woody vegetation and post-burn topkill surveys were conducted at each sample plot in a 3-meter radius. Due to the high abundance of woody shrubs present on the study sites, we tallied woody plants in quadrants, and height and stem density categories to ensure efficient data collection. Within each quadrant, woody species were documented across five height classes (0-0.5m, 0.5-1m, 1-2m, 2-3m, >3m) and by their stem density category (1-25 stems, 26-50 stems, 51-100 stems, > 100 stems). Following each prescribed burn, we also collected data on burn severity at each sample plot by measuring survival and topkill of woody stems. Woody stems were tallied by stem type (topkilled or survived) across the same five height categories by stem density category. Woody stems were considered topkilled if the entire aboveground stem was dead due to fire and evidence of fire damage was present. To ensure accurate detection of stem types (topkilled or survived), survival and topkill surveys following fall and summer burns took place in the next growing season. Surveys following spring burns occurred at minimum 4 weeks after burns or last-frost date.

During post-burn data collection, we also examined each plot for evidence of fire activity to determine whether there was an occurrence of fire at the plot. Evidence of fire activity included topkill due to fire, scorch, and/or char present on the woody and herbaceous plants present. We documented occurrence of fire as a binary variable (yes = activity of fire present, no = no fire activity present).

In addition to monitoring severity of the prescribed burns through vegetation surveys, we conducted biomass collections before and after burns to measure the amount of biomass consumed by fires. Unfortunately, due to difficulty of accurately predicting when prescribed fires would occur and of access to field sites it was not possible to collect biomass immediately before fires, creating a time lag between pre-fire data collection and occurrence of fire. In addition, the fluctuating water table at our sites made it difficult to consistently sample. For example, if water table was the baseline for vegetation collection and it was high on the day of collection then less material was collected that day than on a day with a low water table. Both of these issues meant that the biomass collections did not result in usable data for testing our hypothesis and is therefore not included in this study.

Statistical analysis

To determine whether burn season influenced the severity of the fire, we conducted our analysis at two scales: burn unit and plot scale. The burn unit scale analysis considered all permanent sample plots in the study regardless of whether evidence of fire was present at the plot, which provided a high-level landscape view of the impact of burn season on fire severity. At plot scale we examined fire severity only

on sample plots where evidence of fire was present, and this method provided an assessment of the more direct impacts of burn season on fire severity.

At the burn unit scale only, we assessed the relationship between burn season (spring, summer, fall) and spatial extent of fire severity across the landscape by looking at the binary response variable for evidence of fire at each sample plot (1 = fire present, 0 = no fire present). We tested for an association by using a Pearson's Chi-squared Test for Independence, and calculated a χ^2 statistic and *p*-value with a significance level set at $p < 0.05$, and calculated odds ratios to measure the likelihood of a plot burning in different seasons. R version 3.5.1 (R Core Team, 2018) was used to carry out statistical analysis, and R package 'questionr' (Barnier et al., 2018) was used for the calculation of odds ratios.

At both the burn unit and plot scale, we used linear mixed effects models to examine burn season influence on severity and included total topkilled stems/m^2 as response variable and burn season (spring, summer, fall) as a fixed predictor variable. Total topkilled stems/m^2 was calculated by taking the difference between total pre-burn stems/m^2 and total survived stems/m^2. Pre-burn stems/m^2 was also added to the models as a covariate to account for the variation in stem densities prior to burns. Mixed effects were either site or point nested in site, and we selected the best model based on the lowest Akaike Information Criterion (Akaike, 1974). R version 3.5.1 (R Core Team, 2018) was used to carry out statistical analysis and mixed models. R package 'nlme' (Pinheiro et al., 2018) provided functions for the development of mixed models. Total topkilled stems/m^2 were reported in least square means (lsm) and standard errors (SE), and significance levels set at $p < 0.05$. R package 'emmeans' (Lenth, 2019) was used to calculate least

square means and to conduct post-hoc Tukey HSD testing. Post-hoc Tukey HSD tests were used to identify significant pairwise differences among burn seasons. R package 'tidyverse' (Wickham, 2017) provided functions for dataset manipulation and figure development.

Results

Of the eight individual prescribed burns completed during the course of the project, four burns were conducted in spring, two burns occurred in fall, and two in summer. Individual prescribed burns varied widely in how many sample plots had evidence of fire present during post-burn surveys (Table 1-1), however, the within season variation was low for spring and summer burns. This was not the case for the two fall burns where one site resulted in evidence of fire at 13 of 16 sample plots and the other site at only 3 of 15 sample plots.

At the burn unit scale, using the Pearson's Chi-squared Test for Independence, we found a highly significant association between burn season and presence of fire at sample plots, χ^2 (2, N = 125) = 38.006, $p < 0.001$. Plots located in spring burn sections had 22.4 times greater odds of having evidence of fire present than plots located in summer burn sections (OR 22.4, 95% CI: 7.7 – 74.5, $p < 0.001$), and 7.6 times greater odds than plots located in fall burns (OR 7.6, 95% CI: 2.7 – 23.1, $p < 0.001$). Plots located in fall burn sections had only 2.9 times greater odds of having evidence of burn present than plots located in summer burns (OR 2.9, 95% CI: 1.03 – 8.9, $p < 0.049$).

At the burn unit scale, we found spring burns resulted in significantly higher total topkilled stems/m^2 compared to fall burns ($p < 0.013$) and to summer burns ($p < 0.001$,

Fig. 1-3). Spring burns resulted in an estimated 74.9% topkill across all permanent sample plots, while fall burns resulted in 45.1% topkill and summer burns resulted in 29.1% topkill (topkill percentages are based ratios of least square means).

At plot scale, where we considered only plots where evidence of fire activity was present, we found no significant differences among burn seasons for total topkilled stems/m^2 (Fig. 1-4). Where fire activity was present on sample plots, spring burns resulted in an estimated 81.4% topkill, fall burns 89.3% topkill, and summer burns 80.6% topkill.

Discussion

Land managers of LBEs conduct prescribed burns with a goal of reducing woody plant density in order to maintain high quality habitat for wildlife, and most prescribed burns are conducted in the spring season. Evidence suggesting that prescribed burns in the growing season might have a greater chance of reducing woody plant density has led land managers to consider expanding prescribed burning outside of the spring season (Buckman, 1964; De Groot & Wein, 2004; Pelc et al., 2011; Ruthven et al., 2003). However, the season in which a fire occurs may influence fire severity outcomes due to fluctuations in fire intensity and rate of spread because of seasonal variation in the herbaceous fuel layer (Whelan, 1995), and could potentially limit the effectiveness of conducting burns in different seasons. The objective of this study was to determine whether burn season led to significant differences in the severity of the fire through analysis of topkilled stems/m^2 and evidence of fire at sample plots. Our findings indicate burn season did have a significant impact on the severity of fire in LBEs. Spring burn

sample plots were more likely to burn than either summer or fall sample plots, and thus at the burn unit scale higher amounts of topkilled stems/m^2 occurred following spring burns. However, where fire did occur, there was no difference in the amount of topkilled stems/m^2 among burn seasons. Thus, the varying impacts of burn season on fire severity appear in the difference in the spatial extent of fire and not as direct fire season effects.

The lethal temperature for death of cambial tissue in woody plants is roughly 60° C, and the amount of time for cambial tissues to reach this lethal temperature is directly related to bark thickness (Whelan, 1995). As bark thickness increases, exposure time to fire increases before mortality of the cambium occurs. However, in thin barked plants, such as those woody shrubs present in LBEs, cambial mortality can occur in a minute or two even in low intensity fires (Stephan et al., 2010). Therefore, a high proportion of shrub topkill should be expected where fire does occur on the landscape within LBEs.

Variation in the fuel load is the most likely reason for the differences in severity at the landscape level and spread of fire (Whelan, 1995). Fuels, in addition to the presence of a heat source and oxygen, are a key element in determining whether fire will ignite and spread (DeBano et al., 1998). Generally speaking, in LBEs, spring prescribed burns occur after snow-melt in April-May, prior to the start of the growing season. As illustrated in a similar study conducted in prairies, herbaceous fuel layers in the spring are typically fully dead and dried with no live herbaceous material present, and fuel moisture is low (Steuter, 1986). Thus, fuels are readily available in the spring in contrast to summer. When summer prescribed burns occur, in August-September, herbaceous plants are near peak growing season and often exhibit higher fuel moisture than would be present in spring or fall (Steuter, 1986). Fall prescribed burns, conducted in October-

November after the first-frost, occur when herbaceous fuels are in the process of curing and fuel moisture is relatively low (Steuter, 1986). Given these general parallels between spring and fall fuel conditions in contrast to summer, we had assumed that fall burns would have similar severity outcomes to spring burns and expected no discrepancy in the likelihood of plots burning between the spring and fall seasons. However, the within season variation in results for fall season prescribed burns suggest that this time of year may result in more inconsistent fuel conditions.

While these assumptions related to fuels may explain the variation present between spring and summer, we did not directly measure factors such as fuel moisture and fuel load continuity. These other factors may have also contributed to the within and between season variation, particularly for the variation between the fall burns. Fuel moisture exerts a strong influence the intensity of the fire, and high moisture content can reduce combustion of herbaceous fuels (DeBano et al., 1998; Whelan, 1995). Moisture levels can vary greatly and are dependent on many factors such as relative humidity, plant canopy, and wind speed (Whelan, 1995). Additionally, prescribed fires in LBEs are surface driven fires carried by the herbaceous fuels, and are therefore dependent on the presence of an herbaceous fuel load to carry fire. However, herbaceous fuel loads in LBEs may vary greatly due to the heterogeneity in the structure in the woody plant overstory and the diversity in the plant community. Areas with a high density and cover of woody plants are more likely to have lower herbaceous fuel loads than those in open areas (Van Auken, 2009), and therefore, could reduce the likelihood of fire in these areas. Continuous fuels are needed for fire to spread effectively, however, dense woody areas within LBEs may act as natural fire breaks (Whelan, 1995). Future research in LBEs

should examine fuel moisture and fuel heterogeneity to assess if variation in fuels explain some of these seasonal differences in fire severity.

Another possibility for the spatial variation in different seasons, may be a result of personnel experience. Given that the majority of prescribed burns occur in the spring season, burn crews may be less familiar with how fires behave in summer and fall. If fires in different seasons ignite and spread differently, it may take burn crews some time to adjust to these differences. As burn crews gain more experience in conducting prescribed burns throughout different seasons, it's likely that they will become more successful in burning in all seasons.

In this study, we examined the effects of conducting prescribed burns in spring, summer, and fall on fire severity. We found that fire severity differed across burn seasons, but only in the spatial extent of the fire. Ultimately, spring burns were the most severe as more area burned during this season, and therefore, more woody stems were topkilled. Where fire was present, there was no difference in the amount of topkilled stems across any of the burn seasons. The exact mechanism for the spatial variation is unknown, but it's likely that seasonal variation in fuel related factors may explain some these results (Whelan, 1995). However, fire behavior is complex and other factors may also be impacting these results such as burn crew familiarity with burning outside of traditional burn seasons. Additional research into the variation in spatial severity is needed in order to determine the root cause of the variation in severity among burn seasons.

Management implications

Assessing the impacts of burn season on severity provides land managers with a better understanding of opportunities to meet long term goals for LBEs. One of the primary managements objectives for conducting prescribed burns is the reduction of woody plants achieved through high severity burns and a high proportion of topkilled stems. During the course of our study, burns conducted in the spring season were more effective than fall or summer at achieving high severity burns and a high proportion of topkill. Therefore, our results suggest that spring prescribed burns provide the best opportunity for meeting these management objectives.

However, prescribed burns conducted in the dormant season may also lead to vigorous resprouting of woody stems in comparison to burns in the growing season (Buckman, 1964; De Groot & Wein, 2004; Pelc et al., 2011; Robertson & Hmielowski, 2014; Ruthven et al., 2003). High amounts of shrub resprouting may lead to little overall change in stem density (Lee et al., 2005). Further study will examine changes in stem density to determine if this is the case, or if the reduction in stems due to high proportion of topkill following spring burns ultimately leads to an overall reduction in woody plant density.

Another potential management strategy for reducing woody plant cover on LBEs is increasing fire frequency. While our study did not test burn frequency impacts, studies have shown that woody plant cover and resprout vigor declines as fire frequency increases (Lee et al., 2005; Peterson et al., 2007; Quinlan et al., 2003). Increasing frequency of fire on the landscape, regardless of fire season, may help deplete energy reserves in root systems (De Groot & Wein, 2004; Janicke & Fick, 1998; Loescher et al., 1990), with little impact to the herbaceous community (Lee et al., 2005; Lewis &

Harshbarger, 1976; Peterson et al., 2007). Thus, to achieve the long-term goal of reducing woody stems in LBEs, land managers may want to consider increasing the frequency of spring burns. Given that spring prescribed burns resulted in a high amount of topkill through achieving the greatest spatial extent of the fire, repeated spring burns may provide the most effective method for reducing woody plants.

However, recent research in Minnesota suggests that mixed levels of burn severity may support the greater bird abundances, compared to high severity fires, due to the patchiness mixed severity fires create on the landscape (Zlonis et al., 2019). Therefore, while our recommendations are provided on the basis that the goal is to uniformly reduce woody plants, we recognize that maintaining heterogeneity for the bird community may also be a management objective, and in those situations, spring burns may not be ideal. Summer and fall burns may be better suited for providing the desired response of mixed-severity burns, and therefore, a varied fire regime across the region is likely the most desirable for managing LBEs in Minnesota.

Table 1-1. Prescribed burns completed at study sites between fall 2016 and spring 2018. Table includes study site, burn unit (season), date when burn was completed (burn date), the total number of permanent sample plots in the burn unit (burn unit *n*), and the number of plots that had evidence of fire present (plot scale *n*).

Study site	Season	Burn date	Burn unit *n*	Plot scale *n*
Hasty Brook	Spring	May 10, 2017	16	16
Highway 29	Spring	May 12, 2017	16	12
Gerzin	Spring	May 16, 2018	16	13
Deer Run	Spring	May 23, 2018	16	16
Highway 29	Summer	Aug 11, 2017	14	4
Hasty Brook	Summer	Sept 12, 2017	16	4
Hasty Brook	Fall	Nov 16, 2016	16	13
Highway 29	Fall	Oct 19, 2017	15	3

Figure 1-1. Study sites (Gerzin, Hwy 29, Deer Run WMA, Hasty Brook) located in northeastern Minnesota within Aitkin, Carlton, and St. Louis counties on both private and public lands designated as Priority Open Landscapes by the MN DNR.

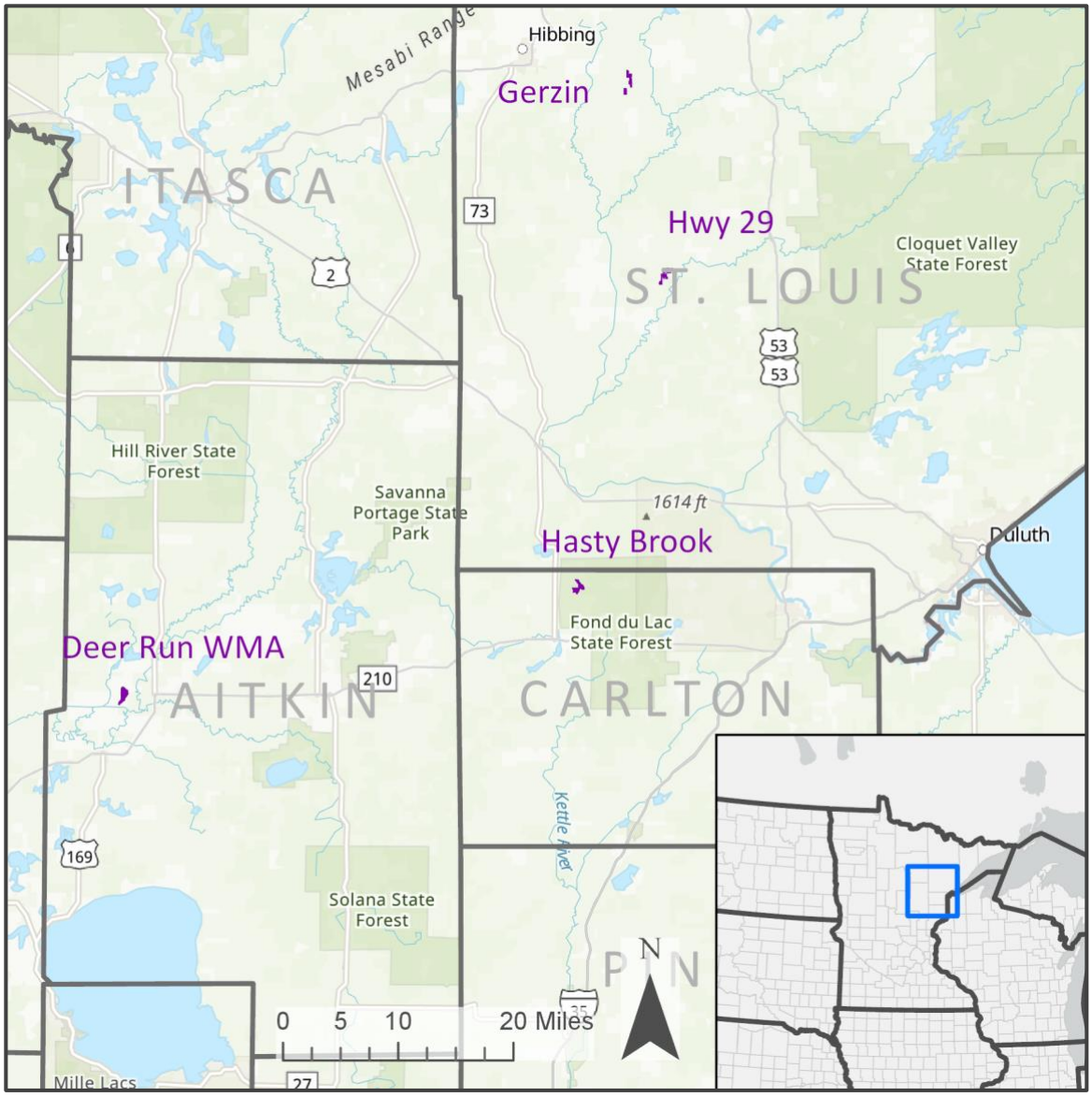


Figure 1-2. Study site design and sample plot layout example from Hasty Brook site located in Carlton county. Each study site included four treatment units (control, spring, summer, fall), each unit approximately 40 hectares and included 16 permanent sample plot locations. Two permanent sample plot locations established near each point count location to support a collaborating study on burn season impacts to avian community.

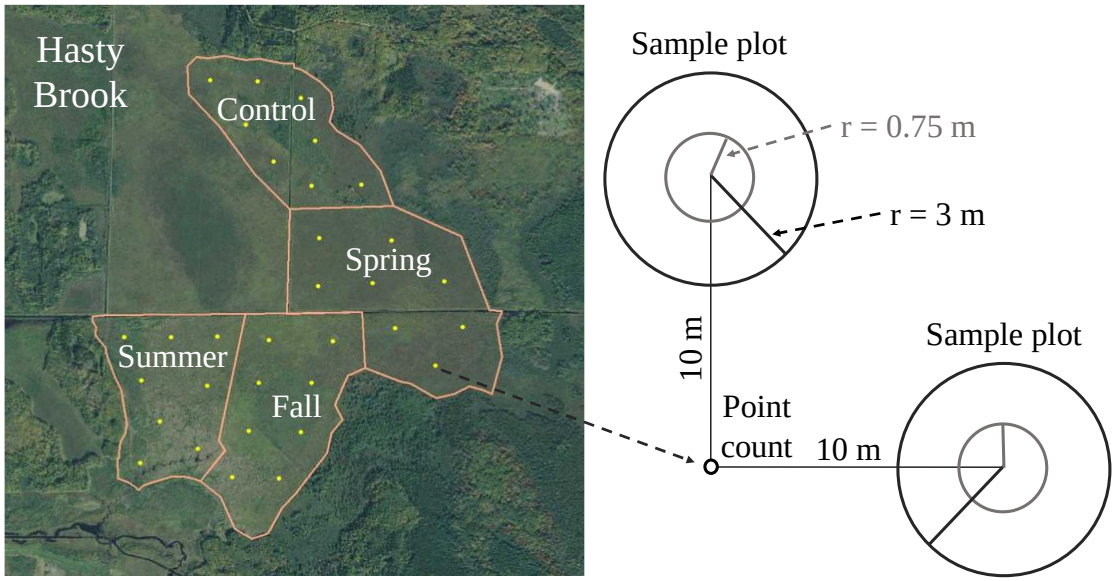


Figure 1-3. Number of topkilled stems (stems/m^2) for prescribed burns executed in three different seasons. Results are at the burn unit scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences between burn seasons among survived stems and among topkilled stems. Sample size (*n*) identifies the number of permanent sample plots included in model for fixed factor (Season). At burn unit scale, all plots were included regardless of the presence fire.

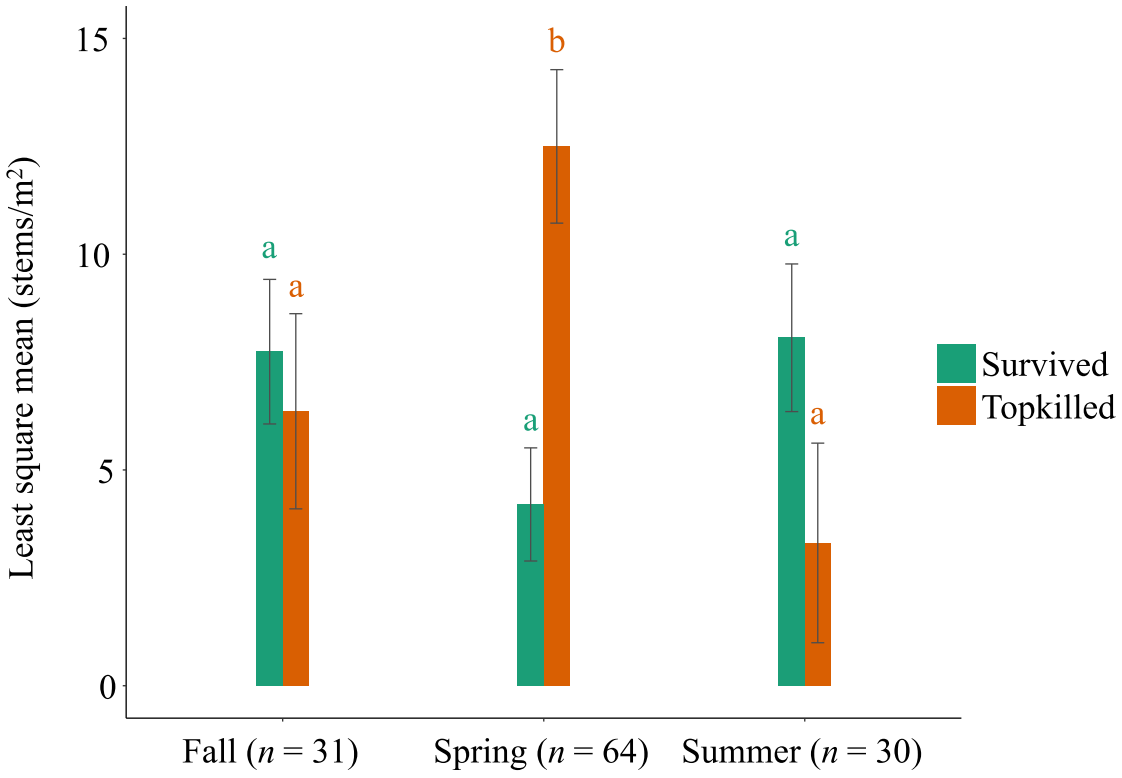
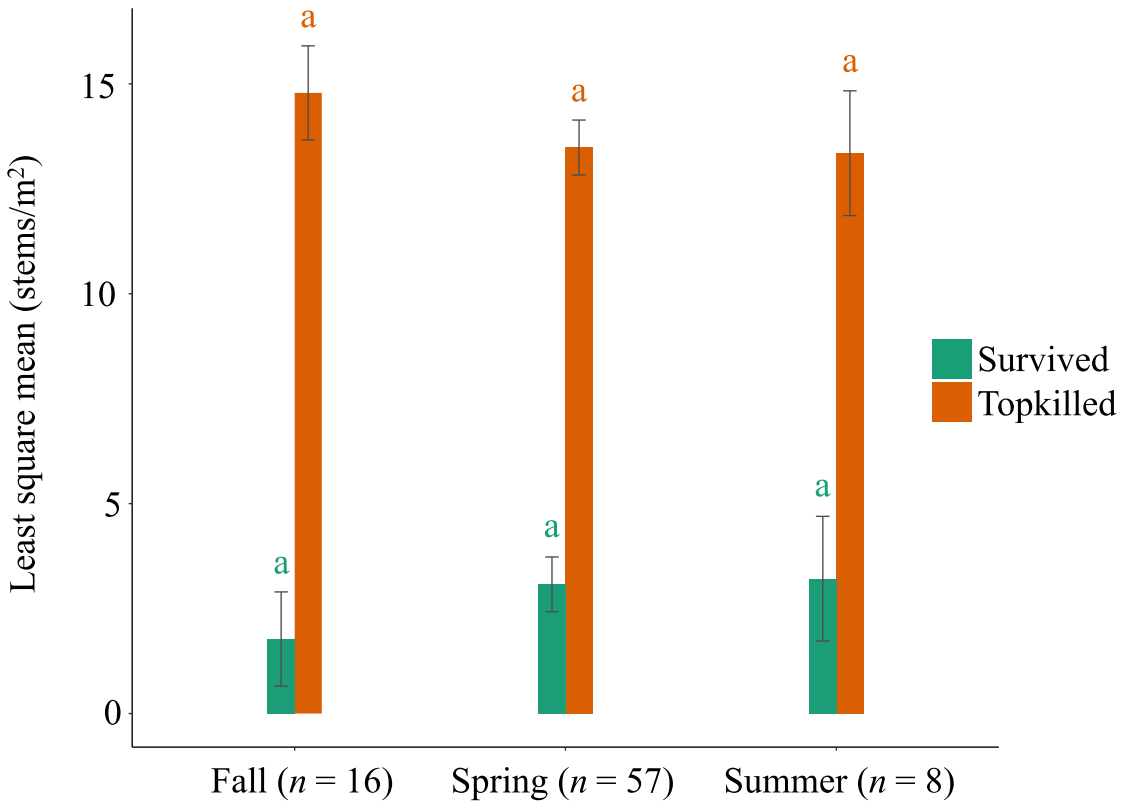


Figure 1-4. Number of topkilled stems (stems/m^2) for prescribed burns executed in three different seasons. Results are at the plot scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences between burn seasons among survived stems and among topkilled stems. Sample size (*n*) identifies the number of permanent sample plots included in model for fixed factor (Season). At plot scale, plots were only included where evidence of fire was detected post-burn.



Chapter 2 Response of herbs and shrubs to seasonal prescribed burning in northern Minnesota lowland brush ecosystems

Introduction

Fire disturbance is as a key component in the conservation of many terrestrial ecosystems (DeBano et al., 1998; Nowacki & Abrams, 2008; Whelan, 1995). Fire can inhibit woody plant encroachment and enhance herbaceous understories (Bart et al., 2016; Curtis, 1959; Vogl, 1969), and can create patchiness to the landscape which increases habitat value for wildlife (Franklin et al., 2007; Lyon et al., 2000). Today, fire is part of a natural disturbance regime resulting from cloud to ground lightning strikes (Pyne, 1982; Zajac & Rutledge, 2001), and a managed disturbance through the use of prescribed fire to maintain fire-dependent plant communities throughout the United States (Ryan et al., 2013).

In Minnesota, land managers of lowland brush ecosystems (LBEs) depend on fire disturbance to maintain critical wildlife habitat (Curtis, 1959; Hanowski et al., 1999). LBEs include a suite of different plant communities such as wet meadow grass and sedge, alder swamps, ericaceous shrub bogs, and willow shrub cars (Curtis, 1959; MN DNR, 2003). These LBE plant communities provide diverse habitat to over 80 Species of Greatest Conservation Need (SGCN), and game wildlife such as sharp-tailed grouse and white-tailed deer (MN DNR, 2015). Without fire, succession in LBEs will often lead to mature, dense and uniform shrub canopies, that shade out the understory herbaceous layer, and reduces the habitat value and the fine fuels needed to carry fire in the future (Curtis, 1959; Hanowski et al., 1999; Ryan et al., 2013).

Currently, natural resource managers use prescribed fire as a management tool in LBEs to reduce woody plant density and maintain LBE wildlife habitat. Sharp tailed grouse, for example, prefer open grassland habitat with few low shrubs present (Hanowski et al., 2000), and fire helps managers maintain these conditions for this species. Prescribed burns in LBEs generally occur in the spring season only. The spring season creates ideal conditions for prescribed burning as herbaceous thatch layers are often at their driest, smoke is more easily managed, and fire conditions are generally safer for burn crews (Knapp et al., 2009). For these reasons, as well as the difficulty in organizing personnel around short and unreliable weather windows, burning outside of the spring season is currently uncommon. However, a prescribed fire regime that relies on spring burning alone may not mimic historic fire regimes, which may limit land managers ability to reduce woody plant density, enhance the herbaceous understory, and ultimately maintain high-quality LBE habitat.

Historically, fire disturbance likely occurred throughout the spring, summer, and fall seasons due to either intentional fire setting by humans (Higgins, 1986), or naturally through lightning strikes (Pyne, 1982; Zajac & Rutledge, 2001). This diversity in fire season may be critical to moderating the woody plant component of LBEs. While there is a lack of research on the impacts of fire season in LBEs, a growing body of research in other ecosystems has shown that the season in which a fire or aboveground disturbance takes place can affect woody resprouting and post-burn recovery (Adams et al., 1982; Brockway & Lewis, 1997; Buckman, 1964; Cronan et al., 2015; De Groot & Wein, 2004; Drewa et al., 2002; Pelc et al., 2011; Robertson & Hmielowski, 2014; Ruthven et al., 2003; Willcox & Giuliano, 2010).

In Southeast US pine forests, growing season burns have been shown to reduce hardwood plant species at a greater rate than dormant season burns (Robertson & Hmielowski, 2014). However, in this same forest type, growing season burns have had mixed success in reducing saw palmetto, which appears to require repeated annual burning in order to control growth (Cronan et al., 2015; Willcox & Giuliano, 2010). In Minnesota, research suggests that the vigor of resprouting Corylus cornuta will be lower following growing season burns compared to dormant season (Buckman, 1964; Pelc et al., 2011). One reason for this variation in resprout vigor may be the cyclical nature of carbohydrate storage in woody plants, as root reserves are generally lower during the growing season and higher in the dormant season (Loescher et al., 1990).

While resprouting following aboveground stem mortality, hereafter called topkill, is a widespread trait shared by many woody plants, and the vigor with which woody plants resprout may vary depending on the timing of topkill event, resprout vigor may also vary by species (Adams et al., 1982; Mallik & Gimingham, 1985; Michielsen et al., 2017; Pausas & Keeley, 2009). A study conducted in temperate grasslands by Michielson et al. (2017) found resprouting to vary among woody species, and therefore, concluded that the ability to control woody plant growth through prescribed burning would largely be dependent on plant community composition. Regardless of the underlying reason for the variation in resprout vigor, evidence suggests that burn season may influence how the LBE woody plant community responds to prescribed fire.

In addition to reducing the woody plant community, prescribed burns in LBEs are often used to enhance the herbaceous understory and improve wildlife habitat. However, research suggests that fire season may also impact diversity and abundance of forbs,

grasses and sedges (Biondini et al., 1989; Bond & van Wilgen, 1996; Towne & Owensby, 1984; Vermeire & Russell, 2018). In tallgrass prairie ecosystems, Towne and Owensby (1984) found that late-spring burns were ideal for increasing abundance of both big bluestem (*Andropogon gerardii*) and Indian grass (*Sorghastrum nutans*), but resulted in decreased perennial forb and sedge abundance. They also found that early-spring and winter burns led to increased perennial forbs and sedges, but a reduced growth in *A. gerardii* and *S. nutans* (Towne & Owensby, 1984). Additionally, in northern mixed prairies, spring and fall prescribed burns increased abundance of forbs while summer burns resulted in a decrease of forb abundance (Biondini et al., 1989; Vermeire & Russell, 2018). The cause for this variation in cover responses may be related to the strain on different plants and plant functional groups when disturbance occurs during the active growing season (Biondini et al., 1989).

Beyond potential impacts to the plant community, season of burning can also impact the spatial extent of fire (Steuter, 1986; Knosalla Chapter 1). A fire behavior study conducted by Steuter (1986) in northern mixed prairies found that fall and spring burns behaved similarly in terms of estimated fire intensity and rate of spread, but during summer burns these measurements decreased significantly leading to a reduction if successfully burned area. This study was paired with another examining the effects of burn season on prairie forbs where they identified that the impacts of burn season were largely dependent on the spatial extent of fire (Biondini et al., 1989). Furthermore, our research in LBEs found that sample plots located in spring burn units were much more likely to have had evidence of fire present than those of summer or fall, signifying that

spring burns burned more area and in turn resulted significantly higher topkilled stems than fall or summer burns (Knosalla Chapter 1).

Given that 1) little research exists on the impacts of fire on LBEs, 2) the current prescribed fire regime is predominantly limited to the spring season, and 3) that burn season may have a measurable impact to the plant community due to seasonal timing of fire and the spatial extent of burns, our objective was to study how seasonal prescribed burning effects the herbaceous and woody plant communities in LBEs in the first growing season after the burn. For all woody plants as well as individual woody species groups, we hypothesized that spring and fall burns would result in no change in stem density due to vigorous resprouting replacing topkilled stems, but that summer burn seasons would experience a decline in stem density due to less vigorous resprouting. We also hypothesized that grass, sedge, and forb cover would not change following spring and fall burns as fires would occur when plants were inactive, but following summer burns, cover of grasses, sedges, and forbs would decline due to disturbance occurring during active growth periods which could stress herbaceous plants.

Methods

Study system

The study area covered a broad range of northern Minnesota lowland brush ecosystems in a suite of different Native Plant Communities (Minnesota Department of Natural Resources, 2003) including northern alder swamp (FPn73), northern poor fen (APn91), and northern wet meadow/carr (WMn82). These plant communities are diverse in the species that are present, but they are similar in that they lack dominant tree

canopies and they have a water table near or above the soil surface throughout the majority of the year (Minnesota Department of Natural Resources, 2003). The species found within these ecosystems include a wide diversity of herbaceous wetland plants including a broad diversity of grasses, sedges, and forbs, and wetland woody species including *Salix petiolaris* Sm., *Salix discolor* Muhl., *Salix pyrifolia* Andersson, *Cornus sericea* L., *Alnus incana* (L.) Moench *ssp. rugosa* (Du Roi) R.T. Clausen, *Rhododendron groenlandicum* Oeder, *Chamaedaphne calyculata* (L.) Moench, and *Betula pumila* L..

The climate of the study area is generally characterized by mild summers with an average July temperature of 19.1°C, cold winters with an average January temperature of -12.4°C, and an average annual precipitation is 67.4 cm (NOAA, 2019). The growing season, defined here as the number of days between the last-frost and first-frost, lasts approximately 110 days throughout the study area, beginning in late May and lasting through mid-September (NOAA, 2019).

Study sites & treatments

The study area for this research included a combination of public and private lands in Aitkin, Carlton, and St. Louis counties in northern Minnesota (Fig. 2-1). This research was conducted in partnership with the Minnesota Department of Natural Resources (MN DNR). In 2016, the MN DNR selected four study sites. Each study site was approximately 160 hectares, and broken down into 4 units approximately 40 hectares each. Units in this study included a control unit where no burns were conducted, and 3 burn units that were treated during a specified season of burn: spring, summer, or fall.

Spring burns occurred in early-Spring (mid-late May) and typically during the dormant season prior to the last-frost. The spring burns represent the traditional prescribed burn season in the lowland brush ecosystems in northern Minnesota. Summer burn season ran from late-June through mid-September, though all summer burns occurred in mid-late August and September during the growing season. Fall burns occurred in October and November, after the first-frost as plants transitioned into the dormant season. Prior to the 2018 field season, burns were conducted at 8 of the 12 possible burn units including 4 spring, 2 summer, and 2 fall burns (Table 2-1).

Data collection

We completed vegetation surveys from July through August of 2016, 2017, and 2018. Within each control and burn unit, we established 8 avian point-count locations to monitor the presence of birds during the breeding season (Hawkinson, 2019). At each point count location, we established two permanent vegetation sample plots, each 10 meters from the point count center to monitor vegetation (Fig. 2-2). In total, each control and burn unit had 16 permanent sample plots for monitoring vegetation (8 avian point counts per burn unit x 2 sample plots per burn unit).

Due to the high abundance of woody shrubs present on the study sites, we tallied woody plants in quadrants, and height and stem density categories to ensure efficient data collection. Within each quadrant, woody species were documented across five height classes (0-0.5m, 0.5-1m, 1-2m, 2-3m, >3m) and by their stem density category (1-25 stems, 26-50 stems, 51-100 stems, > 100 stems).

At each permanent sample plot, we collected woody and herbaceous vegetation data. Woody plant data was collected in a 3-meter radius plot and across five stem height categories (0-0.5 m, 0.5-1 m, 1-2 m, >2 m). To support efficient data collection, woody plant data was collected in quadrants because of the high abundance of shrubs on sample plots. In each quadrant, we identified each woody species present in a height category and estimated stem density via density categories (1-25 stems, 26-50 stems, 51-100 stems, > 100 stems). Herbaceous data was collected in a nested 0.75m radius plot in Braun-Blanquet (1965) percent cover categories (< 1%, 1-5%, 6-25%, 26-50%, 51-75%, 75%).

Additionally, during post-burn sampling, we identified whether fire had been present or not on each permanent sample plot through evidence of woody stem scorch or char, topkill, or herbaceous fuel consumption.

Statistical analysis

Post-burn data collection found that evidence of fire was not present on all permanent sample plots across the burn units in the study, signifying that results of burn season may be dependent on spatial scale similar to Biondini et al. (1989). Therefore, to determine the impacts of burn season on woody stem density and herbaceous cover, we identified three primary pathways of inquiry. First, at the scale of an entire burn unit, we analyzed how burn season impacted the change in woody stem density and herbaceous cover overall using all permanent sample plots within each burn unit regardless of the presence of fire. The burn unit scale provides a broader landscape view of the changes that occurred during each season of burn. Second, at a plot scale, we analyzed how burn

season impacted the change in woody stem density and herbaceous cover only on permanent sample plots where evidence of fire was present. This plot scale view allowed us to directly assess impacts of burn season on changes in woody plant stem density and herbaceous cover. Third, at a "species" scale, we analyzed how burn season impacted stem density and herbaceous cover on permanent sample plots where a particular species, species group, or functional group was present and where evidence of fire was detected post-burn. The "species" scale provides a more detailed examination of the direct impacts of burn season on the most common lowland brush woody species and herbaceous cover.

For all pathways, we used linear mixed effects models to analyze the impacts of burn season on change in woody stem density and herbaceous cover overall and within woody height categories. Models included burn season as a fixed factor and change in stems/m^2 or change in herbaceous cover as the response variable. Woody species scale models did not always include all height categories due to limits related to species growth form. Additionally, species and functional groups were not uniformly present across all sites or burn seasons, therefore, where species were found in only one plot or less within a burn season, we removed that burn season from the model. For the response variable, we calculated the change in stems/m^2 or herbaceous cover at each permanent sample plot using stem density category or herbaceous cover midpoints from both pre-burn and post-burn data collection (change in stems/m^2 = post-burn stems/m^2 – pre-burn stems/m^2, e.g. post-burn 5.3 stems/m^2 – pre-burn 2.5 stems/m^2 = increase of 2.8 stems/m^2). For pre-burn stems/m^2 and herbaceous cover estimates, we used measurements collected from the most recent survey prior to a prescribed burn. For post-burn stems/m^2 and herbaceous cover, we used post-burn woody stem density and herbaceous cover measurements in the first

growing season post-burn. For example, for a spring 2018 prescribed burn, we used pre-burn stem density measurements collected in 2017 and post-burn stem density measurements collected in 2018.

Four woody plant species, two woody plant species groups, and three herbaceous functional groups were selected for our "species" scale analysis. Woody species were selected based upon the most frequently detected woody species across the majority of control and burn units. The four individual woody species included *Cornus sericea* L., *Alnus incana* (L.) Moench *ssp. rugosa* (Du Roi) R.T. Clausen, *Betula pumila* L., and *Spiraea alba* Du Roi. One species group lumped all ericaceous shrubs, including *Rhododendron groenlandicum* Oeder, *Chamaedaphne calyculata* (L.), *Kalmia polifolia* Wangenh., *Andromeda polifolia* L., *Vaccinium angustifolium* Aiton and *V. myrtilloides* Michx.. The other species group lumped all Salix sp., including *Salix petiolaris* Sm., *S. discolor* Muhl., *S. pyrifolia* Andersson, *S. planifolia* Pursh, *S. serissima* (L.H. Bailey) Fernald, *S. bebbiana* Sarg., *S. pedicellaris* Pursh, *S. lucida* Muhl., *S. candida* Flueggé ex Willd., *S. humilis* Marshall, and *S. eriocephala* Michx.. "Species" scale analysis for herbaceous cover was broken down in three broad functional group categories of grasses, sedges, and forbs (forb group also includes ferns and fern allies).

Given the potential for variation among study sites, point count locations, and vegetation sample plots, we used step wise model selection, including random effects of site, point nested in site, and plot nested in point nested in site, and selected the best model based on the lowest Akaike Information Criterion (Akaike, 1974). Significance was determined at $p < 0.05$ and significant pairwise differences between burn seasons were identified using post-hoc Tukey HSD tests. Results were reported in least square

means (lsm) and standard errors (SE), in units of stems/m^2. All statistical analyses were conducted in R version 3.5.1 (R Core Team, 2018), mixed models were developed using the 'nlme' package in R (Pinheiro et al., 2018), significant differences were identified using the 'emmeans' package in R (Lenth, 2019), and the 'tidyverse' package in R (Wickham, 2017) was used to manipulate datasets and create graphical figures.

Results

Pre-burn stem density & cover

Average total pre-burn woody stem density across all burn and control units was 17.9 stems/m^2 (± 0.611). Average pre-burn stem density was greatest in the 0-0.5m height category (6.72 ± 0.507 stems/m2) and decreased with each increase in height category (Fig. 2-3). Across all burn and control units, average grass percent cover was 37.8 (± 2.3), average forb percent cover was 43.6 (± 3.3), and average sedge percent cover was 32.6 (± 2.1).

Burn unit scale

At the burn unit scale (all sample plots included in the models regardless of the presence of fire), we found significant changes in stem density across all height categories for spring burns compared to the control (Fig. 2-4). Overall, total stem density (sum of stems/m^2 in all height categories) increased in the control units, decreased following spring burns, and did not change following fall or summer burns. Measures in total stem density masked changes within height categories that often went in opposing directions.

In the 0-0.5m height category, stems/m^2 increased significantly following spring burns. While in all other height categories (0.5-1m, 1-2m, >2m), stems/m^2 significantly decreased following spring burns. In contrast, the changes in stems/m^2 following both fall and summer burns were not found to be significantly different from controls in any height category.

We found significant pairwise differences among burn seasons in several height categories. In the 0-0.5m height category, we found that stems/m^2 increased following spring and fall burns, and decreased following summer burns. In the 1-2m height category, stems/m^2 decreased following spring burns compared to summer burns.

Plot scale

At the plot scale, considering only those permanent sample plots with evidence of fire, we found similar magnitude and direction of change in stems/m^2 for spring burns across all height categories as we found at the burn unit scale (Fig. 2-5). Following spring burns, stems/m^2 significantly increased in the lowest height category (0-0.5m) and decreased significantly in all other height categories (0.5-1m, 1-2m, >2m). However, we also found significant differences between fall burns and controls that were not detected at the burn unit scale. In the lowest height category (0-0.5m), there was a significant increase in stems/m^2 following fall compared to the control, and a significant decrease in stems/m^2 for fall burns in the 1-2m height category.

Across seasons of burn, we found significant pairwise differences at the plot scale. In the lowest height category (0-0.5m), stems/m^2 decreased in summer burns, while stems/m^2 increased for fall and spring burns. We also continued to find a significant

pairwise difference between the decrease in stems/m^2 in spring burns and no detected change in stems/m^2 in summer burns in the 1-2m height category.

Total stems/m^2 decreased across all seasons of burn, but only spring showed a statistically significant decrease compared to the control.

Species scale

We identified 47 native woody plant species and 1 invasive woody plant species (*Frangula alnus* Mill.), and over 150 native herbaceous species and 17 invasive herbaceous plant species (Table 2-2). The top 10 most frequently detected woody plant species based on occurrence in sample plots on average between 2016-2018 were *Spiraea alba, Salix petiolaris, Betula pumila, Chamaedaphne calyculata, Salix pyrifolia, Cornus sericea, Alnus incana ssp. rugosa, Salix discolor*, *Rhododendron groenlandicum*, and *Salix planifolia*. The top 10 most frequently detected herbaceous plant species were *Calamagrostis canadensis* (Michx.) P. Beauv., *Dryopteris cristata* (L.) A. Gray, *Eutrochium maculatum* (L.) E.E. Lamont, *Campanula aparinoides* Pursh, *Carex lacustris* Willd., *Thelypteris palustris* Schott, *Carex lasiocarpa* Ehrh., *Carex stricta* Lam., *Solidago uliginosa* Nutt., and *Symphyotrichum boreale* (Torr. & A. Gray) Á. Löve & D. Löve. Results are presented on the herbaceous response in broad "species" groups including 41 species in the sedge group, 13 species in the grass group, and 115 species in the forb group (forb group includes ferns and fern allies). Results are presented on the woody response for the top ten woody species as either individual species or "species" groups for Salix spp. and the ericaceous shrubs.

For herbaceous "species" groups, no significant differences in cover change were found between burn season treatments and control units for grasses, sedges, or forbs (Fig. 2-6). However, we found significant pairwise differences among burn seasons in the forb functional group. We found that forb cover increased following spring burns and decreased following summer burns.

For the *Alnus incana ssp. rugosa* model, the summer burn was dropped due to insufficient sample size ($n < 2$). Overall, there were no significant differences in the change in total stems/m^2. However, within height categories, *A. rugosa* stems/m^2 significantly increased in the 0-0.5m height category following both spring and fall burns (Fig. 2-7). In the tallest height categories (1-2m, >2m), we found significant decreases in A. rugosa stems/m^2 in fall burns.

Similarly, the summer burn was dropped from the *Cornus sericea* model due to insufficient sample size ($n < 2$). Additionally, woody stems of *C. sericea* rarely reached heights >2 meters in our study area, and therefore, there are no results available in the >2m height category. Overall, we found that total stems/m^2 of *C. sericea* significantly decreased following spring burns (Fig. 2-8). No significant differences were found in the 0-0.5m height category following spring or fall burns. In the 0.5-1m and 1-2m height categories, we found significant decreases in stems/m^2 following spring burns, and we also found significant decreases in stems/m^2 in the 0.5-1m height category following fall burns.

As with *Cornus sericea*, woody stems of *Spiraea alba* rarely reached heights >2 meters in our study area, and therefore, there are no results available in >2m height category. In the 1-2m height category we found Spiraea alba stems/m^2 decreased

significantly following spring and fall burns compared to control (Fig. 2-9). However, neither of these decreases in stems/m^2 was significantly different than summer. No significant differences were found in the two lowest height categories (0-0.5m, 0.5-1m), or for overall total change in stems/m^2 for Spiraea alba.

Salix spp. were prevalent throughout all burn seasons and height categories, therefore, no modifications to the species scale model was required for the *Salix spp*. group. Following spring and fall burns, we found that stems/m^2 of *Salix spp*. increased significantly in the 0-0.5 height category, and decreased significantly in the 1-2m height category (Fig. 2-10). In addition, spring burns decreased significantly in the >2m height category. No significant change in *Salix spp*. stems/m^2 was detected for summer burns in any height category. However, we found a pairwise difference in the 0-0.5m height category where stems/m^2 increased following fall burns and did not change following summer burns.

No significant differences were found among the burn seasons and the control for change in stems/m^2 for *Betula pumila* or the ericaceous shrub species group in any height category (Fig. 2-11 & Fig. 2-12). At plots where they occurred, ericaceous shrubs were detected in very high abundances during our vegetation monitoring, but their presence was limited to spring and control units and to the 0-0.5m and 0.5-1m height categories.

Discussion

Prescribed burns are widely used as a management tool to decrease woody plant density and enhance the herbaceous plant community. The current prescribed fire regime in many ecosystems is generally limited to the dormant season (Knapp et al., 2009).

However, prescribed burns in different seasons can result in a wide range of effects on woody plant density across ecosystems depending on resprout vigor and topkill (Adams et al., 1982; Brockway & Lewis, 1997; Cronan et al., 2015; Lee et al., 2005; Willcox & Giuliano, 2010). Many factors have been found to impact resprout vigor following aboveground disturbance including the size and age of individual plants, and the severity of the disturbance (Bond & Midgley, 2001; Bond & van Wilgen, 1996). However, seasonal variation in carbohydrate storage may also impact the vigor of woody resprouts after a topkill event (Buckman, 1964; De Groot & Wein, 2004; Pelc et al., 2011; Ruthven et al., 2003). Additionally, resprout vigor has also been shown to vary by woody plant species (Adams et al., 1982; Bond & Midgley, 2001; Mallik & Gimingham, 1985; Michielsen et al., 2017). Thus, changes in woody plant density following prescribed burns may be a result of season of prescribed burn, but the changes may differ by species or species group. Dormant and growing season burns may also result in variation in response from the herbaceous plant community (Biondini et al., 1989; Bond & van Wilgen, 1996; Lovell et al., 1982; Towne & Owensby, 1984; Vermeire & Russell, 2018). Given the lack of research in LBEs, our objective was to determine whether season of prescribed burning led to significant changes in the woody and herbaceous plant community. Our findings indicate that burn season did result in significant changes in woody plant density; however, the magnitude and direction of detected changes varied by spatial scale (burn unit or plot scale) and "species" scale. Our hypothesis speculating about the effects of burn season on the woody plant community was only partially supported: at the plot scale, fall burns resulted in vigorous resprouts replacing topkilled stems; however, stem density declined following spring burns and did not change for

summer burns. At the burn unit scale, the effects of burn season following fall and summer burns were masked due to the large number of unburned sample plots indicating that burn season impacts may be spatially dependent. Our hypothesis about the herbaceous response to burn season was also partially supported: burn season did not impact cover of grasses, sedges, or forbs following burns in any season compared to control units, although spring burns increased forb cover compared to summer burns.

Spatial extent of fire

Both spatial scales (burn unit and plot scale) provide important perspectives on the overall impact of burn season on woody plant density. Where there were a large number of plots that did not have evidence of fire present, such as in the fall and summer burn treatments, changes in overall stem density were muted at the burn unit scale. Thus, while our results indicate that burn season treatments led to significant differences in stem density change, the magnitude of burn season impacts varied by spatial scale. Our findings are consistent with those of Biondini et al. (1989) which also found prescribed burn impacts dependent on both season and spatial extent of burns. In a collaborating study, Steuter (1987) identified the cause for this variation in spatial scale to be lower fire intensity and rate of spread during summer burns compared to fall and spring burns, and it is likely that our results are similarly influenced. While many factors effect fire intensity and rate of spread, those related to fuel can exert the strongest influence and can also vary greatly across seasons (DeBano et al., 1998; Scott & Burgan, 2005; Whelan, 1995). In surface fire systems, such as LBEs, increased fuel moisture and live herbaceous vegetation found during the growing season can decrease fire intensity and rate of spread

across a landscape (DeBano et al., 1998; Whelan, 1995). Thus, while our results at the burn unit scale indicate that spring burns resulted in the greatest reduction in woody plant density in LBEs, we must be mindful that our findings are scale dependent, and also consider the direct impacts of burn season on the woody plant community through the plot scale results.

Overall woody burn season effects

At the plot scale, the impacts of burn season on woody stem density are no longer masked by the large number of unburned plots as these were removed from the analysis. Our findings of increased stem density in the 0-0.5m height category following fall and spring burns, suggests vigorous resprouting was likely occurring for both burn seasons. These findings are consistent with research that has found that vigorous resprouting of woody plants occurs following dormant season burns (Buckman, 1964; Drewa et al., 2002; Lee et al., 2005; Pelc et al., 2011; Robertson & Hmielowski, 2014; Willcox & Giuliano, 2010).

On the other hand, growing season burns have been shown to result in a reduction in resprout vigor compared to dormant season (Buckman, 1964; Drewa et al., 2002; Pelc et al., 2011; Robertson & Hmielowski, 2014), and our results for summer burns are also consistent with these findings. Following summer burns in LBEs, we found no change in stem density in the 0-0.5m height category, which suggests that little resprouting occurred following growing season burns. Resprouting is a common trait exhibited by many woody plant species (Bond & Midgley, 2001; Pausas & Keeley, 2009), and one reason for variation in resprout vigor may be due to the cyclical nature of carbohydrate

storage in root systems between dormant and growing seasons ((De Groot & Wein, 2004; Janicke & Fick, 1998; Loescher et al., 1990). Our findings may signal the importance of carbohydrate storage cycles in this ecosystem and the potential impact burn season could have on woody plant density in LBEs.

While total stem density fully recovered to pre-burn levels following fall burns at the plot scale, the decrease in total stem density following spring burns may indicate lower levels of carbohydrate storage during the spring burn season. The two fall burns in the study occurred in late-October and mid-November, well after the first frost and into the dormant season. Carbohydrate storage in root systems of woody plants generally peaks in the fall after senescence, but decreases over the winter (Loescher et al., 1990). On the other hand, the four spring burns all occurred in mid-late May. Last frost for the region of our study generally occurs in late-May or early-June, and therefore, our spring prescribed burns were taking place about 2 weeks before the average start of the growing season. Carbohydrate reserves generally begin to deplete even before budbreak as woody plants enter the growing season, moving carbohydrates from roots into aboveground tissues (Loescher et al., 1990), which may align with the timing of our mid-May prescribed burns. Thus, while fall burns occurred when root carbohydrate reserves were likely at their highest, spring burns would have likely occurred when reserves were lower, and could be a reason for the overall decrease in total woody stem density following spring burns while fall burn density recovered to pre-burn levels.

"Species" scale burn season effects

Variation in the seasonal timing of prescribed burns has been shown to result in changes in herbaceous plant cover (Biondini et al., 1989; Lovell et al., 1982; Sparks et al., 1998; Towne & Owensby, 1984; Vermeire & Russell, 2018), and summer burns may be particularly detrimental to some herbaceous species and species groups due to burns occurring during active growing cycles (Biondini et al., 1989; Vermeire & Russell, 2018). The lack of cover change in any burn season for our herbaceous "species" groups compared to the control units suggests that LBEs are relatively unaffected by burns in any season at the broad "species" group level. Our analysis looked at "species" groups encompassing many different species in each (41 sedges, 13 grasses, 115 forbs), and therefore, it is possible that an increase or decrease in one species would have been recovered by another without a change being detected for the overall group. Towne and Owensby's (1984) research in tallgrass prairie considered impacts at the species level of the most common grass species in addition to broad groups, and they found that as one grass declined as a result of spring burning another increased in cover. Additional study should be conducted at the individual species level to further understand the impacts of burn season on LBE herbaceous species.

Many woody species bear the fire adaptive trait of resprouting following topkill (Pausas & Keeley, 2009; USDA NRCS, 2019), but woody species vary in their resprouting response (Adams et al., 1982; Mallik & Gimingham, 1985; Michielsen et al., 2017; Pausas & Keeley, 2009). For the common woody LBE species, the amount of resprouting in response to topkill also appeared to differ by species, and our study demonstrates the impact of burn season on these individual woody species and species groups.

A. rugosa and the *Salix spp.* group are both regarded as prolific resprouters following a topkill event (Brisson et al., 2006; Lee et al., 2005; Newsholme, 1992; USDA NRCS, 2019), and our findings strongly support this claim, particularly following fall and spring burns. In *Salix caroliniana* Michx. dominated wetlands in Florida, single dormant season burns resulted in no change in stem density as *Salix* resprouting recovered stems lost to topkill in the first growing season (Lee et al., 2005). Our study found similar results for both *A. rugosa* and the *Salix spp.* group following spring and fall burns, where significant increases in the 0-0.5m height category exceeded the decreases in taller stems, and ultimately, there was no overall change in total stem density. As the summer season was removed from the analysis for *A. rugosa*, we are unable to determine the impacts of growing season burns on this species. However, our results for summer burns for the *Salix spp.* group align with prior studies that growing season topkill may help reduce the vigor of resprouting (Buckman, 1964; De Groot & Wein, 2004; Pelc et al., 2011; Ruthven et al., 2003).

While both *S. alba* and *C. sericea* are known to resprout (USDA NRCS, 2019), and studies suggest that *C. sericea* in particular has high resprouting capabilities following fire disturbance (Gordon, 1976; Smith & James, 1978), our study did not find evidence of this in LBEs. Our findings suggest that even with declines in stem density in taller height categories, there was overall less resprouting occurring following topkill in any burn season for these species. However, we had to remove the summer burn season data from the analysis for *C. sericea*, and are therefore, unable to determine growing season impacts. Given the significant decreases in stem density in taller height categories for *C. sericea* and *S. alba*, if vigorous resprouting was occurring, we would have

expected to see a corresponding increase in stem density in the 0-0.5m height category. One possible reason for our results for *C. sericea* was the sub-dominant nature of individuals of the species present in LBEs. While *C. sericea* can grow to heights greater than 3 meters throughout much of its range (USDA NRCS, 2019), it was rare for this species to grow above 2 meters in the LBE study sites. This could mean that *C. sericea* experiences overall reduced vigor in LBEs and is therefore limited in its ability to resprout prolifically post-fire. There appears to be very little research on how fire disturbance impacts *S. alba* beyond identifying that *S. alba* can resprout after topkill. It is therefore likely that our study is unique to understanding burn season impacts to the species. Additionally, the standard error of some least square mean estimates for S. alba may indicate greater variation in mortality and resprouting than we have found for some other common LBE species.

Neither *B. pumila* nor the ericaceous shrub group exhibited significant differences in stem density changes among any of the burn seasons in LBEs. However, similar to *S. alba*, little research exists on the effects of fire on *B. pumila*. De Groot and Wein (2004) found that burn season influences resprouting of *Betula glandulosa* Michx., a closely related species which can hybridize with *B. pumila*. In their study, early season burns led to higher numbers of resprouting stems than burns conducted later in the season. While our results for *B. pumila* do not align with their findings for *B. glandulosa*, our results may also be limited by a small sample size for fall (n=4) and summer (n=2). Our study results for the LBE ericaceous shrub group included a number of species such as *Chamaedaphne calyculata*, *Rhododendron groenlandicum*, and *Vaccinium angustifolium* and *V. myrtilloides*. In a study of the effects of seasonal fire in an Acadian forest bog with

a similar species combination, researchers found that spring and fall fires led to high resprouting of species within the ericaceous shrub group (Flinn & Wein, 1988). While our results may not support these findings by Flinn and Wein (1988), our results were limited to data collected following spring burns only, a fairly low sample size, and data analysis challenges related to our data collection methods where high stem densities were present. Additional research and seasonal burning in LBEs in follow-up to this study will help to support this analysis on the impacts of burn season on *B. pumila* and the ericaceous shrub group species.

While common LBE species varied in their responses to burn season at the plot scale, the direct impact of fires in different seasons suggests that woody plants may quickly recover to pre-burn stem density. In the first growing season post-burn, only spring burns resulted in decreased stem density in both the burn unit and plot scales. Additional years of monitoring of these study sites will help improve our understanding of the impacts of burn season. Furthermore, studies have also demonstrated that frequency of fire may be equally if not more important than burn season in reducing woody plant density on the landscape (Lee et al., 2005; Pelc et al., 2011; Pendergrass et al., 1998; Peterson et al., 2007; Quinlan et al., 2003). Lee et al. (2005) found that single dormant season burns in Salix spp. wetlands in Florida resulted in high resprouting and no change in stem density; however, a second dormant season burn led to a decrease Salix spp. stem density. Given that frequency of topkill has been found to result in a reduction in resprout vigor for many woody plant species, additional research should be conducted in LBEs to help determine if frequency in addition to burn season is a viable method for controlling woody plant encroachment in LBEs.

Management implications

Reduction in woody encroachment and enhancement of the herbaceous plant community are primary objectives for prescribed burning in LBEs where the current prescribed burn regime favors spring burning. Based on our results, spring burns are currently the most effective at reducing overall woody stem density. However, there was no evidence that spring burns, or burns in any season, enhanced the herbaceous plant community. Regardless, our results identify the current spring burn timing as the optimal season for burning to meet the identified management objective for woody vegetation. However, the end result of spring burning may reduce heterogeneity on the landscape by creating a uniform cohort of regenerating woody understory, which recent research suggests may reduce habitat quality for the bird community (Zlonis et al., 2019). Thus, land managers may want to consider alternatives to the spring season to increase patchiness.

Our results following fall and spring burns were consistent at the plot scale when evidence of fire was present. This suggests that fall burns, unlike spring, may leave more areas untouched by fire, and therefore increase woody heterogeneity across the landscape, which may increase habitat quality for the bird habitat quality (Zlonis et al., 2019). Ultimately, given that the impacts of burn season to woody stem density are dependent on the spatial extent of the fire, land managers should consider burn season as a tool to help achieve their desired outcomes in LBEs for density as well as patchiness across the landscape.

Although our study did not look at the impacts of burn frequency in addition to burn season, land managers may want to consider if increasing fire frequency could further accomplish management objectives given that resprout vigor may decline as fire frequency increases (Lee et al., 2005; Peterson et al., 2007; Quinlan et al., 2003). It is possible that burn frequency coupled with season could further management objectives related to reducing woody density as well increasing heterogeneity. Ultimately, prescribed burn season should be viewed as a tool for supporting a variety of outcomes in LBEs, and burn season selection should depend on the desired effects for a specific site.

Table 2-1. Prescribed burns completed at study sites between fall 2016 and spring 2018. Table includes study site, burn unit (season), date when burn was completed (burn date), the total number of permanent sample plots in the burn unit (burn unit *n*), and the number of plots that had evidence of fire present (plot scale *n*).

Study site	Season	Burn date	Burn unit *n*	Plot scale *n*
Hasty Brook	Spring	May 10, 2017	16	16
Highway 29	Spring	May 12, 2017	16	12
Gerzin	Spring	May 16, 2018	16	13
Deer Run	Spring	May 23, 2018	16	16
Highway 29	Summer	Aug 11, 2017	14	4
Hasty Brook	Summer	Sept 12, 2017	16	4
Hasty Brook	Fall	Nov 16, 2016	16	13
Highway 29	Fall	Oct 19, 2017	15	3

Table 2-2. Woody and herbaceous species detected throughout study sites before and after prescribed burns in 2016, 2017, and 2018. Species listed by scientific name in alphabetical order within category and "species" scale groups. **I**=Non-native species.

Category & "Species" Scale Group	Species	Common name
Herbaceous – Forbs, Ferns, and Fern Allies		
	Achillea millefolium L.	common yarrow
	Amphicarpaea bracteate (L.) Fernald	American hogpeanut
	Anaphalis margaritacea (L.) Benth.	pearly everlasting
	Anemone canadensis L.	Canadian anemone
	Anemone quinquefolia L.	wood anemone
	Apocynum androsaemifolium L.	spreading dogbane
	Aralia nudicaulis L.	wild sarsaparilla
	Asclepias incarnate L.	swamp milkweed
	Asclepias syriaca L.	common milkweed
	Athyrium filix-femina (L.) Roth	common ladyfern
	Bidens connata Muhl. ex Willd.	purplestem beggarticks
	Botrychium oneidense (Gilbert) House	bluntlobe grapefern
	Calla palustris L.	wild calla
	Caltha palustris L.	marsh marigold
	Campanula aparinoides Pursh	marsh bellflower
	Cerastium arvense L.	field chickweed
	Cerastium fontanum Baumg	common mouse-ear chickweed (**I**)
	Chamerion angustifolium (L.) Holub	fireweed
	Chelone glabra L.	white turtlehead
	Cicuta bulbifera L.	bulblet-bearing water hemlock
	Cirsium arvense (L.) Scop.	Canada thistle (**I**)
	Cirsium muticum Michx.	swamp thistle
	Clintonia borealis (Aiton) Raf.	bluebead
	Comarum palustre L.	Marsh Cinquefoil
	Coptis trifolia (L.) Salisb.	threeleaf goldthread
	Cornus canadensis L.	bunchberry
	Doellingeria umbellata (Mill.) Nees	flat-topped white aster
	Drosera rotundifolia L.	round-leaved sundew
	Dryopteris carthusiana (Vill.) H.P. Fuchs	spinulose woodfern
	Dryopteris cristata (L.) A. Gray	crested fern
	Epilobium leptophyllum Raf.	bog willowherb
	Equisetum arvense L.	field horsetail
	Equisetum fluviatile L.	water horsetail
	Equisetum palustre L.	marsh horsetail
	Equisetum pretense Ehrh.	meadow horsetail
	Equisetum sylvaticum L.	woodland horsetail
	Eupatorium perfoliatum L.	common boneset
	Eurybia macrophylla (L.) Cass.	large-leaved aster

Euthamia graminifolia (L.) Nutt.	grass-leaved goldenrod
Eutrochium maculatum (L.) E.E. Lamont	spotted joe pye weed
Polygonum scandens L. var. *scandens*	climbing false buckwheat
Fragaria vesca L.	woodland strawberry
Fragaria virginiana Duchesne	wild strawberry
Galium asprellum Michx.	rough bedstraw
Galium labradoricum (Wiegand) Wiegand	labrador bedstraw
Galium obtusum Bigelow	bluntleaf bedstraw
Galium tinctorium (L.) Scop.	stiff marsh bedstraw
Galium trifidum L.	threepetal bedstraw
Gentiana andrewsii Griseb.	bottle gentian
Gentiana rubricaulis Schwein.	Great Lakes gentian
Geum rivale L.	purple avens
Halenia deflexa (Sm.) Griseb.	American spurred gentian
Hieracium aurantiacum L.	orange hawkweed (**I**)
Hieracium scabrum Michx.	sticky hawkweed
Hieracium umbellatum L.	narrow-leaf hawkweed
Impatiens capensis Meerb.	spotted touch-me-not
Iris versicolor L.	blueflag iris
Lactuca serriola L.	prickly lettuce
Lathyrus ochroleucus Hook.	cream pea
Leucanthemum vulgare Lam.	oxeye daisy (**I**)
Lycopus americanus Muhl. ex W.P.C. Barton	American water horehound
Lycopus uniflorus Michx.	northern bugleweed
Lysimachia ciliata L.	fringed loosestrife
Lysimachia terrestris (L.) Britton, Sterns & Poggenb.	swamp candles
Lysimachia thyrsiflora L.	tufted loosestrife
Maianthemum canadense Desf.	Canada mayflower
Maianthemum trifolium (L.) Sloboda	threeleaf false Solomon's-seal
Malaxis unifolia Michx.	green adder's-mouth orchid
Mentha arvensis L.	wild mint
Onoclea sensibilis L.	sensitive fern
Ophioglossum pusillum Raf.	northern adderstongue
Osmundastrum cinnamomeum L.	cinnamon fern
Packera aurea (L.) Á. Löve & D. Löve	golden ragwort
Parnassia palustris L.	marsh grass of Parnassus
Persicaria amphibia (L.) Gray p.p.	swamp smartweed
Persicaria sagittata (L.) Gross.	arrowleaf tearthumb
Petasites frigidus (L.) Fr. var. *palmatus* (Aiton) Cronquist	arctic sweet coltsfoot
Petasites frigidus (L.) Fr. var. *sagittatus* (Banks ex Pursh) Cherniawsky	arrowleaf sweet coltsfoot
Platanthera lacera (Michx.) G. Don	green fringed orchid
Platanthera psycodes (L.) Lindl.	lesser purple fringed orchid
Potentilla norvegica L.	rough cinquefoil (**I**)
Prunella vulgaris L.	self-heal

Pteridium aquilinum (L.) Kuhn	bracken
Pyrola americana Sweet	American wintergreen
Pyrola elliptica Nutt.	shinleaf
Ranunculus acris L.	tall buttercup (**I**)
Ranunculus hispidus Michx.	hispid buttercup
Rudbeckia hirta L.	blackeyed Susan
Rumex orbiculatus A. Gray var. *borealis* Rech. f.	greater water dock
Scutellaria galericulata L.	marsh skullcap
Scutellaria lateriflora L.	mad-dog skullcap
Solidago canadensis L.	Canada goldenrod
Solidago gigantea Aiton	giant goldenrod
Solidago uliginosa Nutt.	bog goldenrod
Sonchus arvensis L.	perennial sowthistle (**I**)
Stachys hispida Pursh	hairy hedge nettle
Stellaria longifolia Muhl. ex Willd.	longleaf starwort
Symphyotrichum boreale (Torr. & A. Gray) Á. Löve & D. Löve	northern bog aster
Symphyotrichum leave (L.) Á. Löve & D. Löve	smooth blue aster
Symphyotrichum lanceolatum (Willd.) G.L. Nesom	panicled aster
Symphyotrichum lateriflorum (L.) Á. Löve & D. Löve	calico aster
Symphyotrichum puniceum (L.) Á. Löve & D. Löve	purple-stemmed aster
Symphyotrichum urophyllum (Lindl.) G.L. Nesom	arrowleaf aster
Taraxacum officinale F.H. Wigg.	common dandelion (**I**)
Thelypteris palustris Schott	northern marsh fern
Triadenum fraseri (Spach) Gleason	Fraser's marsh St. Johnswort
Trientalis borealis Raf.	starflower
Trifolium aureum Pollich	golden clover (**I**)
Trifolium hybridum L.	alsike clover (**I**)
Trifolium pratense L.	red clover (**I**)
Trifolium repens L.	white clover (**I**)
Vaccinium oxycoccos L.	small cranberry
Vicia americana Muhl. ex Willd.	American vetch
Vicia sativa L.	common vetch (**I**)
Viola spp.	violet
Herbaceous - Grasses	
Agrostis perennans (Walter) Tuck.	upland bentgrass
Agrostis scabra Willd.	rough bentgrass
Bromus ciliatus L.	fringed brome
Calamagrostis canadensis (Michx.) P. Beauv.	Canada bluejoint
Elymus repens (L.) Gould	quackgrass (**I**)
Glyceria canadensis (Michx.) Trin.	rattlesnake mannagrass
Glyceria grandis S. Watson	American mannagrass

	Glyceria striata (Lam.) Hitchc.	fowl mannagrass
	Muhlenbergia glomerata (Willd.) Trin.	spiked muhly
	Phalaris arundinacea L.	reed canarygrass (**I**)
	Phleum pratense L.	timothy (**I**)
	Poa palustris L.	fowl bluegrass
	Poa pratensis L.	Kentucky bluegrass (**I**)
Herbaceous – Sedges		
	Carex arctata Boott ex Hook.	drooping wood sedge
	Carex aurea Nutt.	golden-fruited sedge
	Carex bebbii Olney ex Fernald	Bebb's sedge
	Carex brunnescens (Pers.) Poir.	brownish sedge
	Carex buxbaumii Wahlenb.	Buxbaum's sedge
	Carex canescens L.	silvery sedge
	Carex castanea Wahlenb.	chestnut sedge
	Carex chordorrhiza Ehrh. ex L. f.	creeping sedge
	Carex crawfordii Fernald	Crawford's sedge
	Carex disperma Dewey	softleaf sedge
	Carex echinata Murray	star sedge
	Carex gracillima Schwein.	graceful sedge
	Carex hirtifolia Mack.	hairy wood sedge
	Carex interior L.H. Bailey	inland sedge
	Carex intumescens Rudge	greater bladder sedge
	Carex lacustris Willd.	lake sedge
	Carex lasiocarpa Ehrh.	wiregrass sedge
	Carex leptalea Wahlenb.	bristlystalked sedge
	Carex lucorum Willd. ex Link	Blue Ridge sedge
	Carex magellanica Lam.	boreal bog sedge
	Carex oligosperma Michx.	few-seeded sedge
	Carex prairea Dewey ex Alph. Wood	prairie sedge
	Carex projecta Mack.	necklace sedge
	Carex scoparia Schkuhr ex Willd.	broom sedge
	Carex stipata Muhl. ex Willd.	awlfruit sedge
	Carex stricta Lam.	tussock sedge
	Carex tenera Dewey	quill sedge
	Carex trisperma Dewey	three-seeded sedge
	Carex utriculata Boott	common yellow lake sedge
	Carex vesicaria L.	blister sedge
	Carex vulpinoidea Michx.	fox sedge
	Eriophorum angustifolium Honck.	tall cottongrass
	Eriophorum chamissonis C.A. Mey.	russet cottongrass
	Eriophorum vaginatum L.	tussock cottongrass
	Juncus dudleyi Wiegand	Dudley's rush
	Juncus tenuis Willd.	path rush
	Juncus vaseyi Engelm.	Vasey's rush
	Luzula multiflora (Ehrh.) Lej.	common woodrush
	Scirpus atrovirens Willd.	dark green bulrush
	Scirpus cyperinus (L.) Kunth	woolgrass
	Typha latifolia L.	broadleaf cattail

Woody		
	Abies balsamea (L.) Mill.	balsam fir
	Acer rubrum L.	red maple
	Alnus incana (L.) Moench ssp. *rugosa* (Du Roi) R.T. Clausen	speckled alder
	Amelanchier spp.	serviceberry
	Aronia melanocarpa (Michx.) Elliott	black chokeberry
	Betula papyrifera Marshall	paper birch
	Betula pumila L.	bog birch
	Cornus racemose Lam.	gray dogwood
	Cornus sericea L.	redosier dogwood
	Corylus cornuta Marshall	beaked hazelnut
	Frangula alnus Mill.	glossy buckthorn (**I**)
	Ilex verticillata (L.) A. Gray	common winterberry
	Larix laricina (Du Roi) K. Koch	tamarack
	Lonicera hirsute Eaton	hairy honeysuckle
	Lonicera oblongifolia (Goldie) Hook.	swamp fly honeysuckle
	Lonicera villosa (Michx.) Schult.	mountain fly honeysuckle
	Picea glauca (Moench) Voss	white spruce
	Picea mariana (Mill.) Britton, Sterns & Poggenb.	black spruce
	Pinus strobus L.	white pine
	Populus balsamifera L.	balsam poplar
	Populus tremuloides Michx.	quaking aspen
	Prunus virginiana L.	chokecherry
	Rhamnus alnifolia L'Hér.	alderleaf buckthorn
	Ribes spp.	current species
	Rosa spp.	rose species
	Rubus idaeus L.	American red raspberry
	Rubus pubescens Raf.	dwarf red blackberry
	Rubus spp.	raspberry species
	Spiraea alba Du Roi	white meadowsweet
	Spiraea tomentosa L.	steeplebush
	Viburnum lentago L.	nannyberry
Woody – ericaceous shrubs		
	Andromeda polifolia L.	bog rosemary
	Chamaedaphne calyculata (L.) Moench	leatherleaf
	Kalmia polifolia Wangenh.	bog laurel
	Rhododendron groenlandicum (Oeder) K.A. Kron & W.S. Judd	bog Labrador tea
	Vaccinium angustifolium Aiton	lowbush blueberry
	Vaccinium myrtilloides Michx.	velvetleaf huckleberry
Woody – Salix spp.		
	Salix bebbiana Sarg.	Bebb willow
	Salix candida Flueggé ex Willd.	sageleaf willow
	Salix discolor Muhl.	pussy willow
	Salix eriocephala Michx.	heart-leaved willow
	Salix humilis Marshall	prairie willow

Salix lucida Muhl.	shining willow
Salix pedicellaris Pursh	bog willow
Salix petiolaris Sm.	meadow willow
Salix planifolia Pursh	diamond-leaf willow
Salix pyrifolia Andersson	balsam willow
Salix serissima (L.H. Bailey) Fernald	autumn willow

Figure 2-1. Study sites (Gerzin, Hwy 29, Deer Run WMA, Hasty Brook) located in northeastern Minnesota within Aitkin, Carlton, and St. Louis counties on both private and public lands designated as Priority Open Landscapes by the MN DNR.

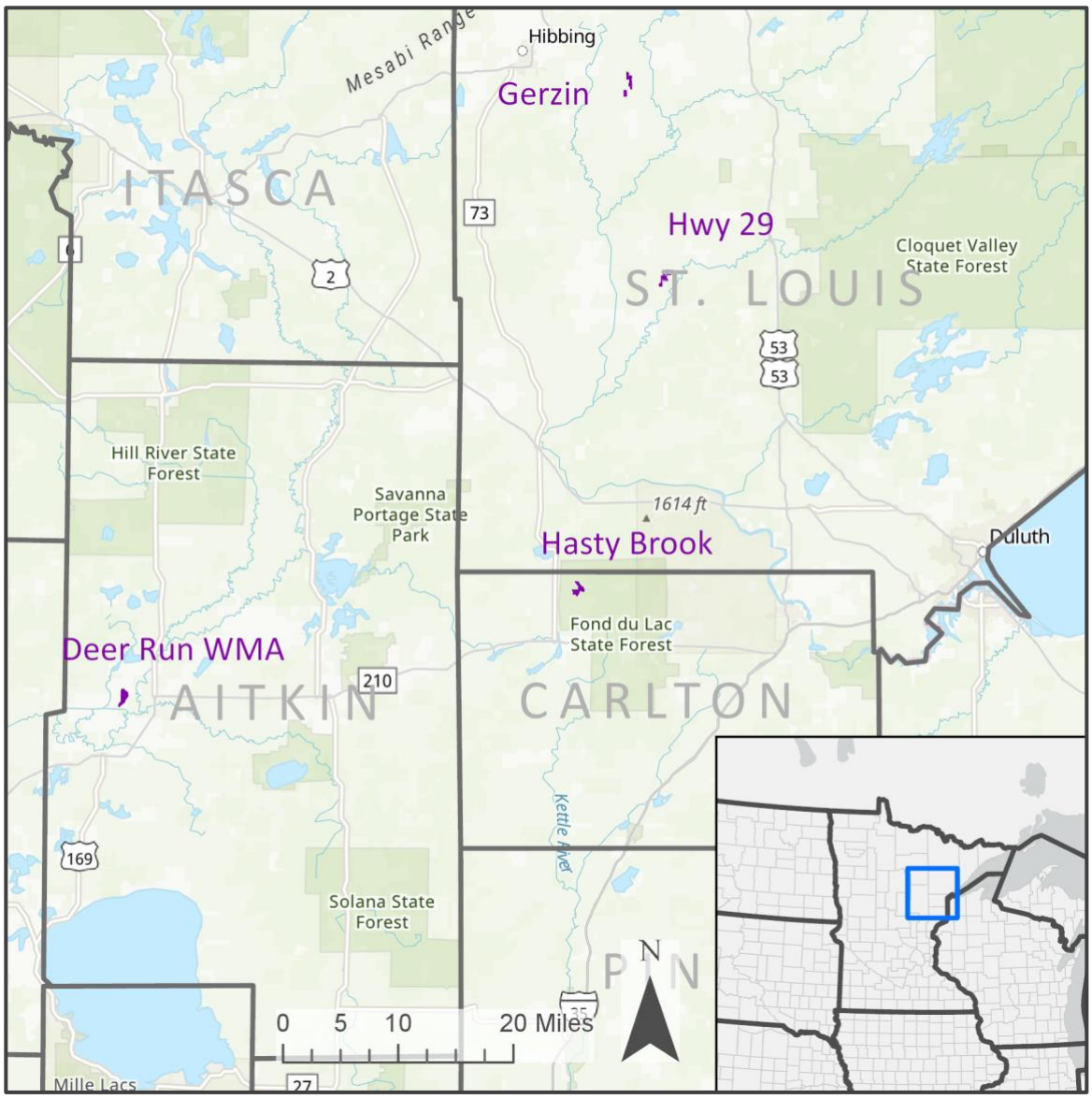


Figure 2-2. Study site design and sample plot layout example from Hasty Brook site located in Carlton county. Each study site included four treatment units (control, spring, summer, fall), each unit approximately 40 hectares and included 16 permanent sample plot locations. Two permanent sample plot locations established near each point count location to support a collaborating study on burn season impacts to avian community.

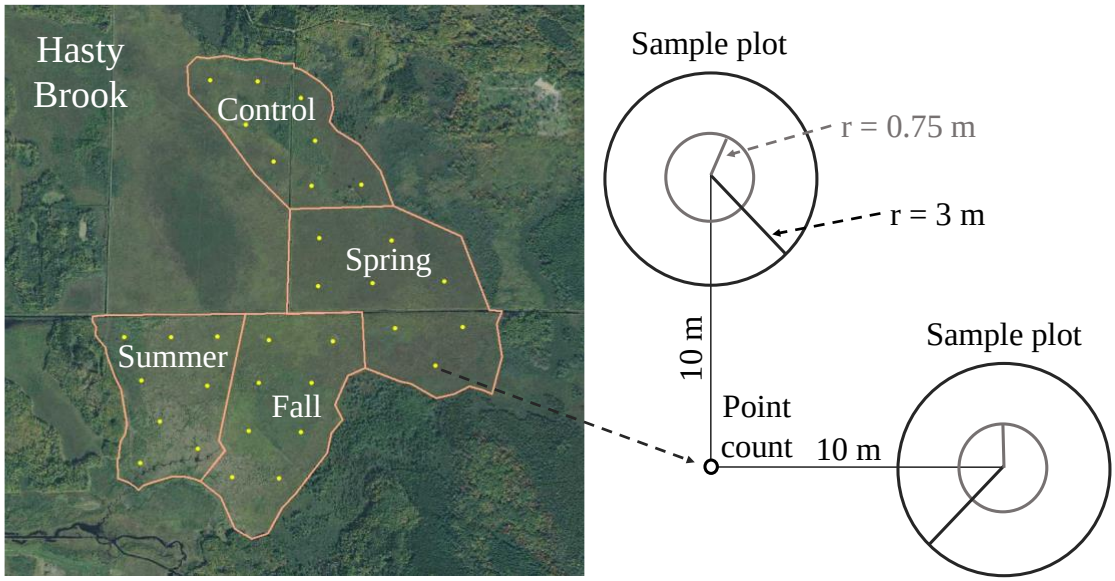


Figure 2-3. Mean pre-burn woody stem density (stems/m^2) and standard errors by height category for all burn and control units. Mean stem density calculated by averaging density across all sample plots (n = 189).

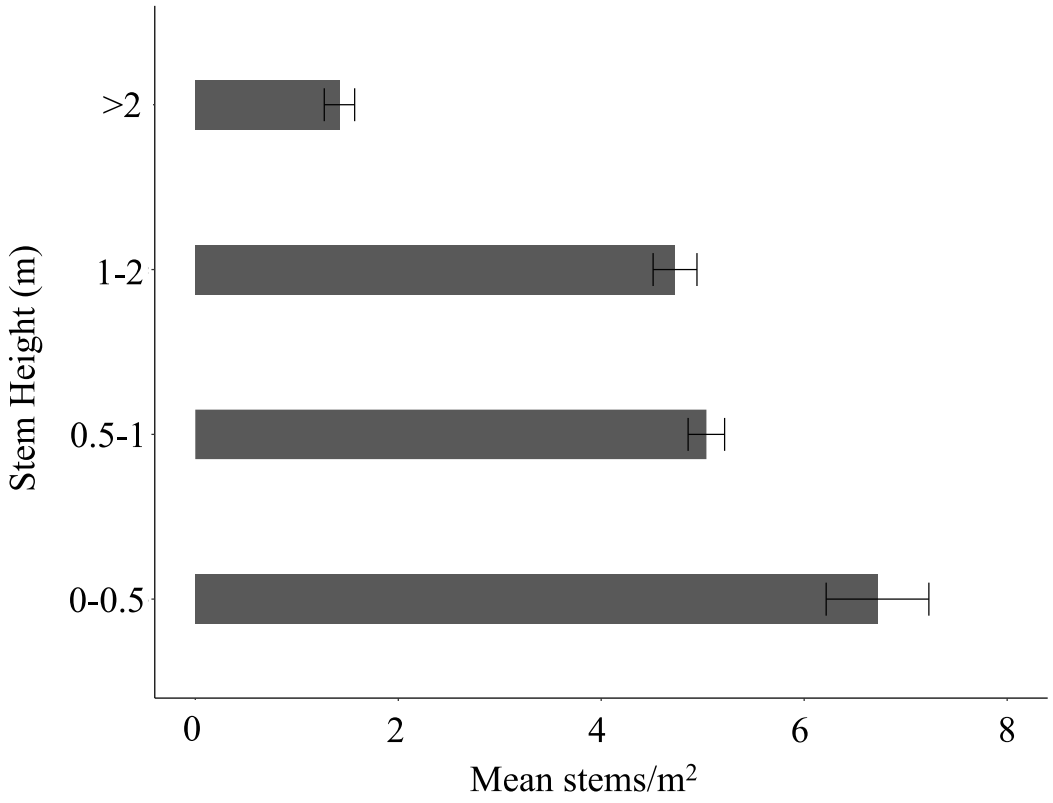


Figure 2-4. Change in woody stem density (stems/m^2) for prescribed burns executed in three different seasons and control unit where no burn occurred. Results are at the burn unit scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences among control and burn seasons in the change in stem density within each height category. Sample size (*n*) identifies the number of permanent sample plots included in model for fixed factor (Season). At burn unit scale, all plots were included regardless of the presence fire.

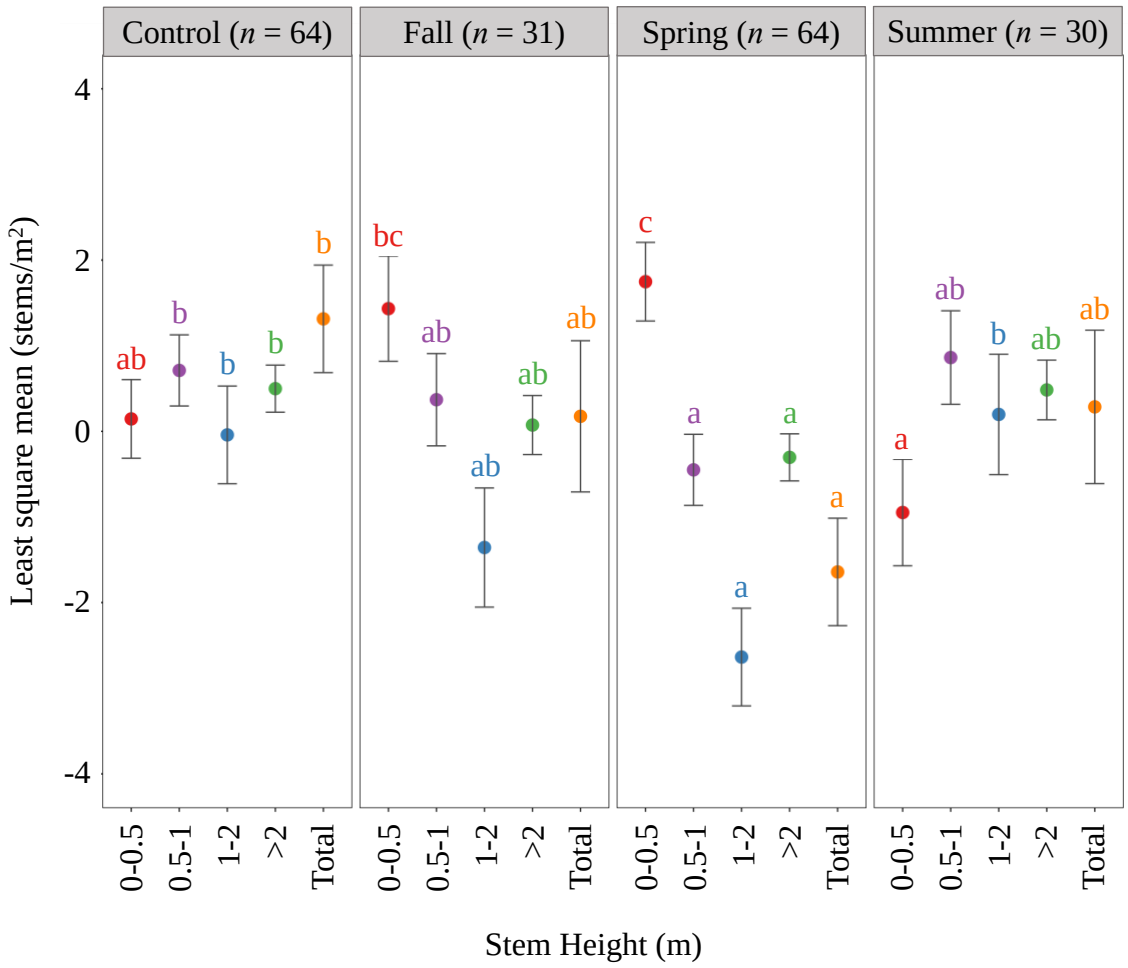


Figure 2-5. Change in woody stem density (stems/m^2) for prescribed burns executed in three different seasons and control unit where no burn occurred. Results are at the plot scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences among control and burn seasons in the change in stem density within each height category. Sample size (*n*) identifies the number of permanent sample plots included in the model for fixed factor (Season). At plot scale, plots were only included where evidence of fire was detected post-burn.

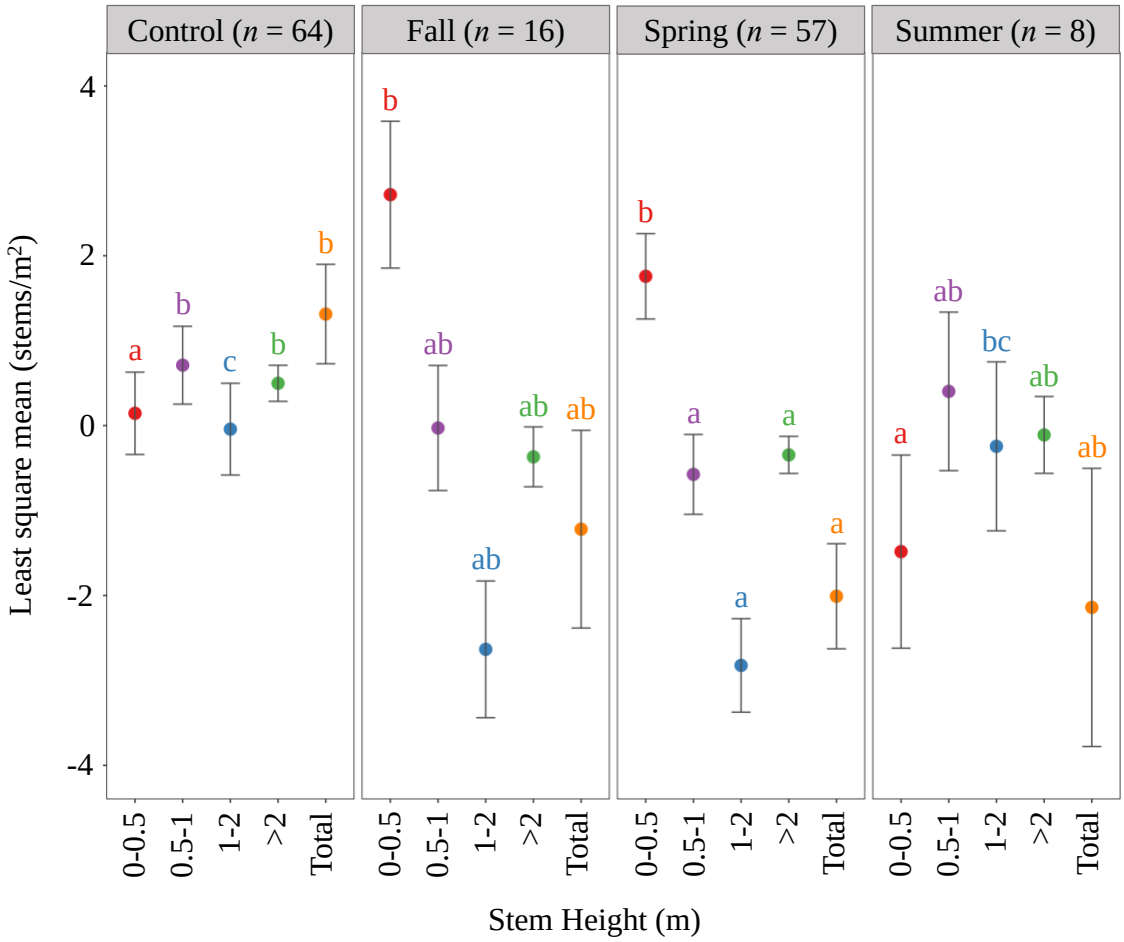


Figure 2-6. Change in herbaceous functional group (forb, grass, sedge) cover (%) for prescribed burns executed in three different seasons and control unit where no burn occurred. Results are at the "species" scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences among control and burn seasons in the change in cover (%) within each herbaceous functional group. Sample size (*n*) identifies the number of permanent sample plots included for each herbaceous functional group in the model for fixed factor (Season). At "species" scale, plots were included where evidence of fire was detected post-burn and the herbaceous functional group was present pre-burn and/or post-burn.

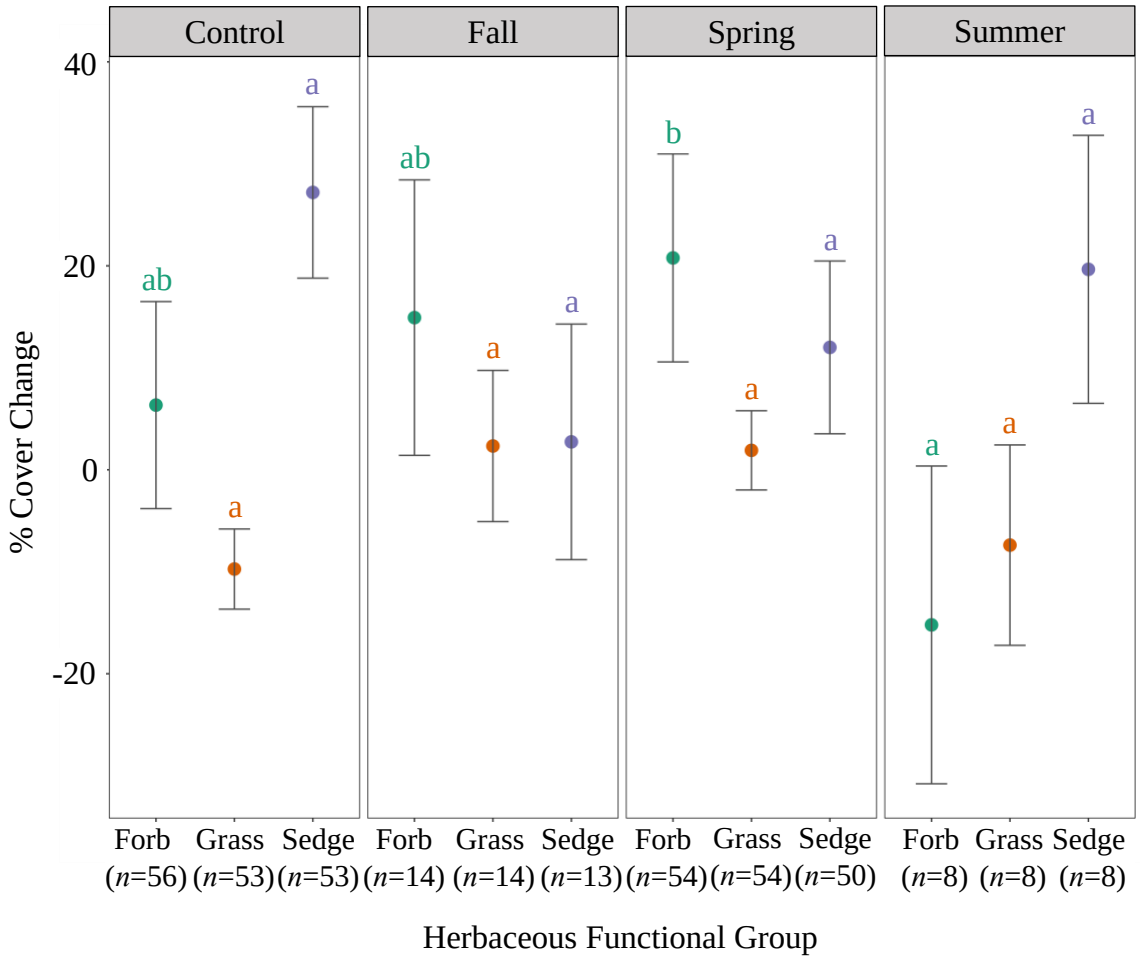


Figure 2-7. Change in *Alnus incana subsp. rugosa* stem density (stems/m^2) for prescribed burns executed in three different seasons and control unit where no burn occurred. Results are in "species" scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences among control and burn seasons in the change in stem density within each height category. Sample size (*n*) identifies the number of permanent sample plots included for fixed factor (Season). At "species" scale, plots were included where evidence of fire was detected post-burn and *Alnus incana subsp. rugosa* was present pre-burn and/or post-burn. The summer burn was not included as a fixed factor due to insufficient sample size (*n* <2).

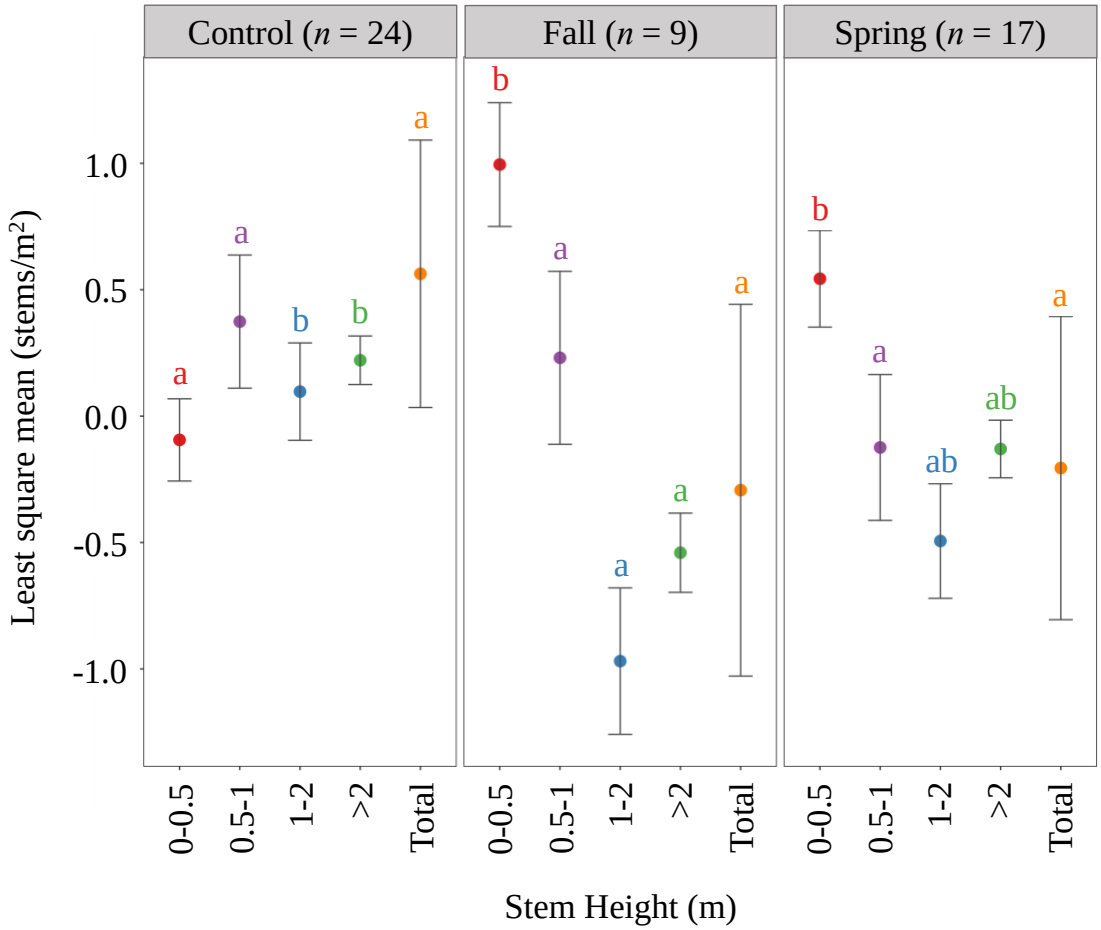


Figure 2-8. Change in *Cornus sericea* stem density (stems/m^2) for prescribed burns executed in three different seasons and control unit where no burn occurred. Results are in "species" scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences among control and burn seasons in the change in stem density within each height category. Sample size (*n*) identifies the number of permanent sample plots included for fixed factor (Season). At "species" scale, plots were included where evidence of fire was detected post-burn and *Cornus sericea* was present pre-burn and/or post-burn. The summer burn was not included as a fixed factor due to insufficient sample size (n<2), and the >2m height category is not included due to infrequency of stems detected above this height in our study area.

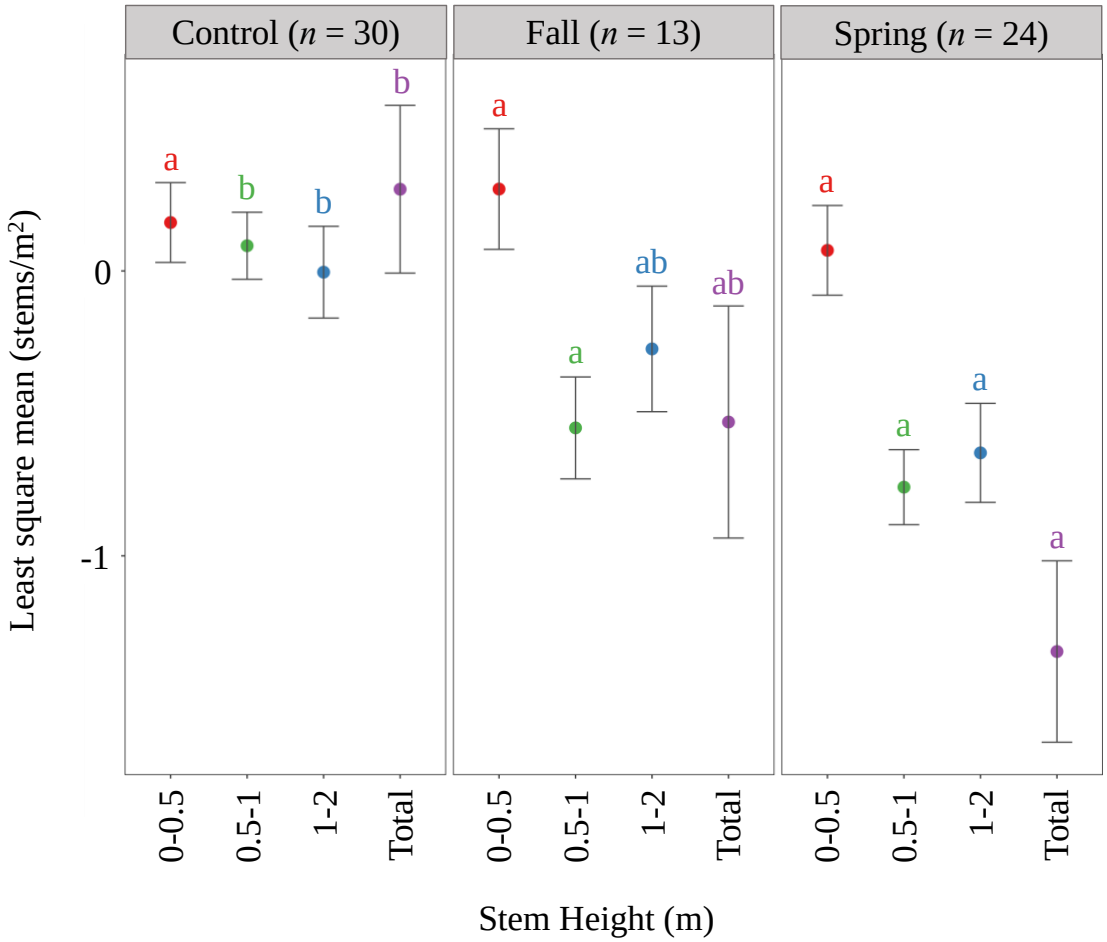


Figure 2-9. Change in *Spiraea alba.* stem density (stems/m^2) for prescribed burns executed in three different seasons and control unit where no burn occurred. Results are in "species" scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences among control and burn seasons in the change in stem density within each height category. Sample size (*n*) identifies the number of permanent sample plots included for fixed factor (Season). At species scale, plots were included where evidence of fire was detected post-burn and *Spiraea alba* was present pre-burn and/or post-burn. The >2m height category is not included due to infrequency of stems detected above this height in our study area.

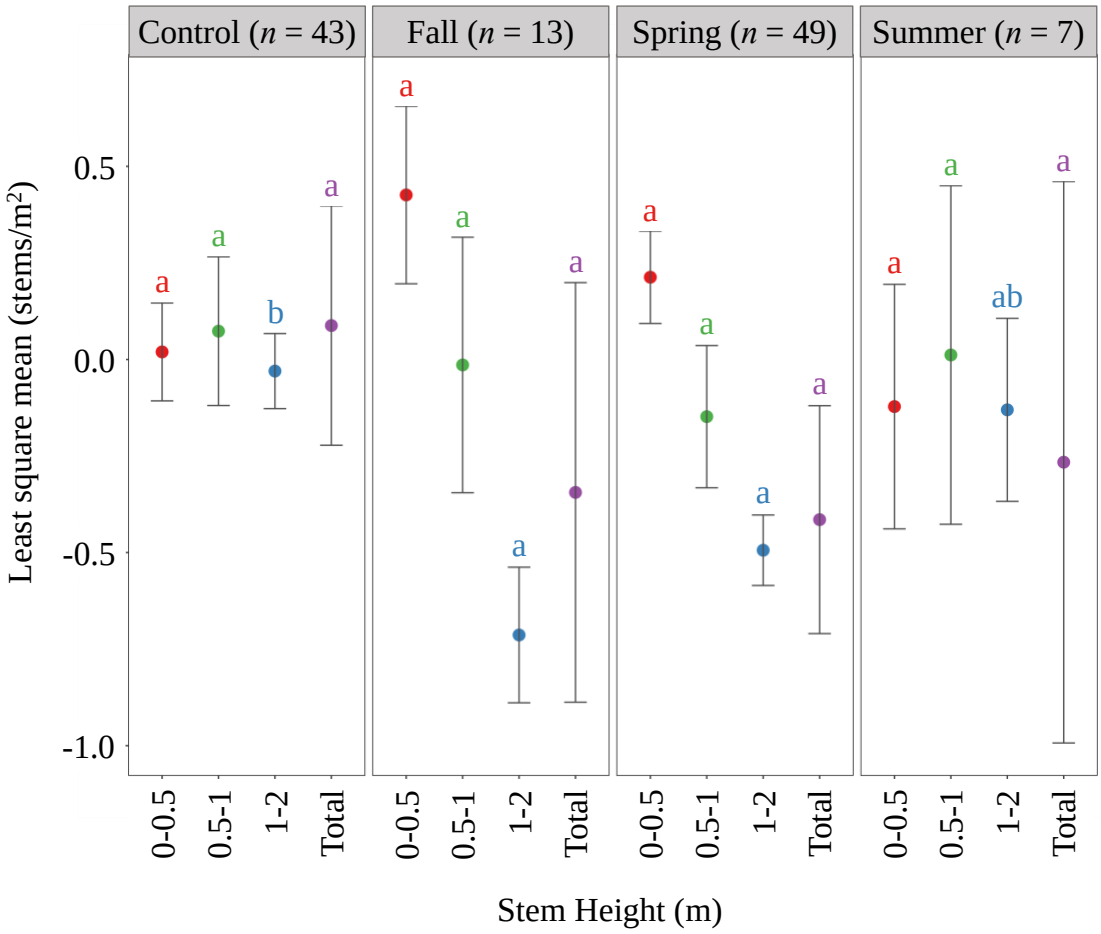


Figure 2-10. Change in *Salix spp.* stem density (stems/m^2) for prescribed burns executed in three different seasons and control unit where no burn occurred. Results are in "species" scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences among control and burn seasons in the change in stem density within each height category. Sample size (*n*) identifies the number of permanent sample plots included for fixed factor (Season). At species scale, plots were included where evidence of fire was detected post-burn and *Salix spp.* were present pre-burn and/or post-burn.

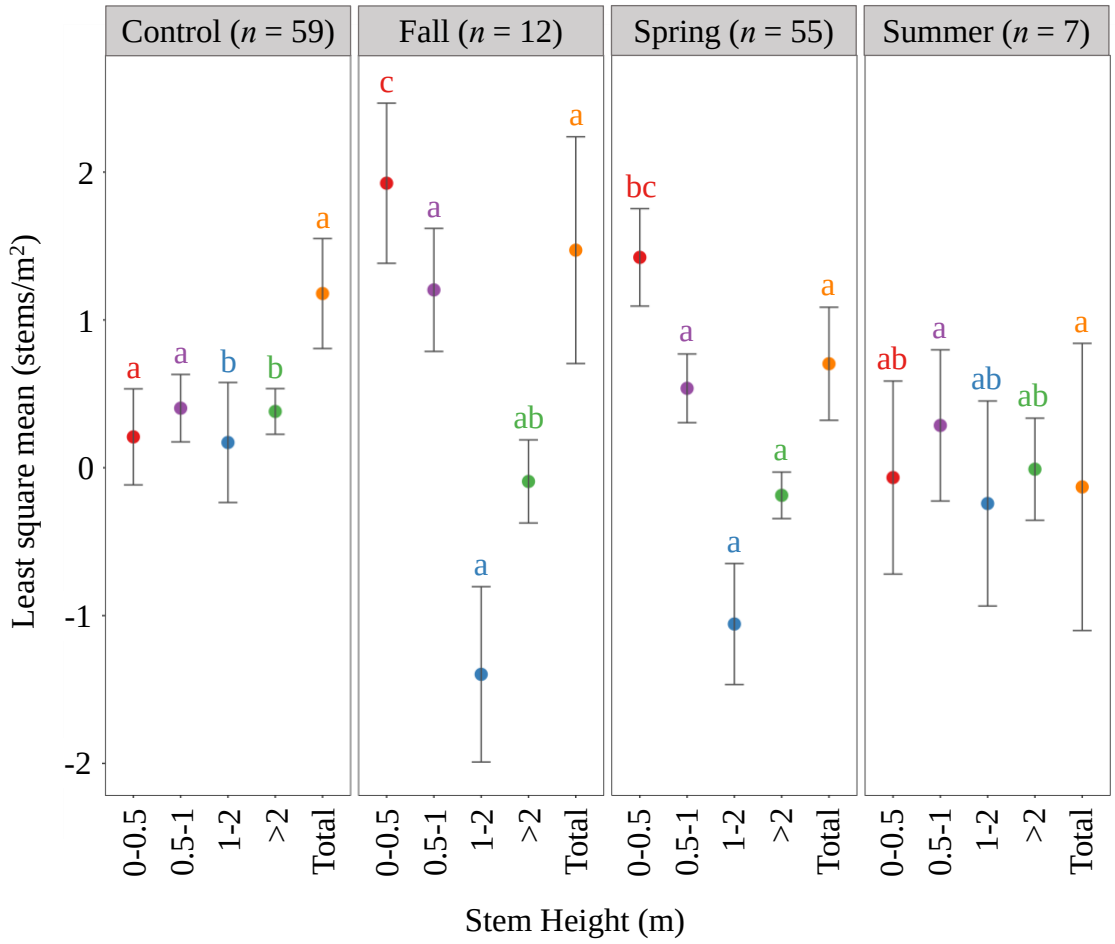


Figure 2-11. Change in *Betula pumila* density (stems/m^2) for prescribed burns executed in three different seasons and control unit where no burn occurred. Results are in "species" scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences among control and burn seasons in the change in stem density within each height category. Sample size (*n*) identifies the number of permanent sample plots included for fixed factor (Season). At species scale, plots were included where evidence of fire was detected post-burn and *Betula pumila* was present pre-burn and/or post-burn.

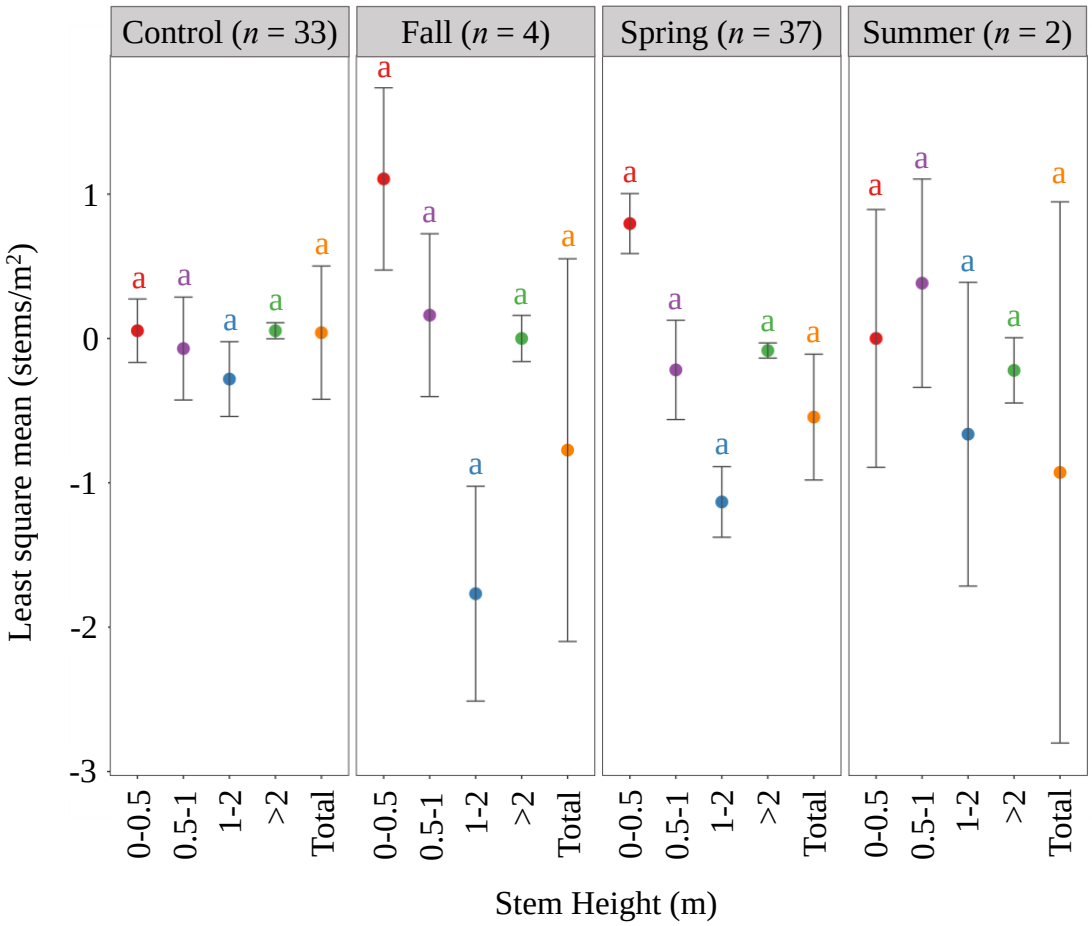
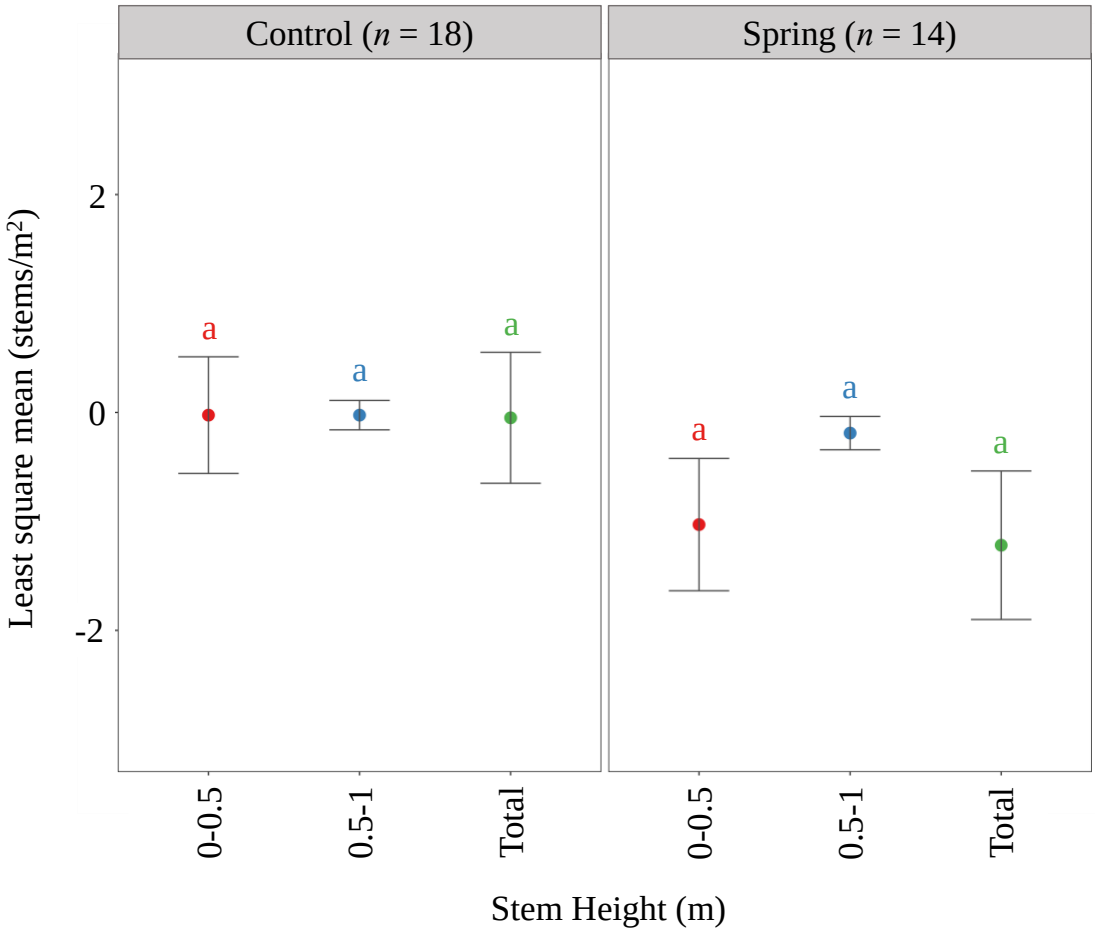


Figure 2-12. Change in ericaceous shrubs stem density (stems/m^2) for prescribed burns executed in three different seasons and control unit where no burn occurred. Results are in "species" scale and reported in least square means and standard errors. Lowercase letters indicate significant pairwise differences among control and burn seasons in the change in stem density within each height category. Sample size (*n*) identifies the number of permanent sample plots included for fixed factor (Season). At "species" scale, plots were included where evidence of fire was detected post-burn and ericaceous shrubs were present pre-burn and/or post-burn. The 1-2m and >2m height categories were not included due to max growth height typically below 1m for ericaceous shrubs in our study area.



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